Quantitative Environmental Analysis, LLC
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Expert Report

Illinois River Watershed Water Quality and Source Assessment

Prepared for:

Illinois River Watershed Joint Defense Group

Prepared by:

Quantitative Environmental Analysis, LLC

Montvale, NJ

January 30, 2009

UNITED STATES DISTRICT COURT
FOR THE NORTHERN DISTRICT OF OKLAHOMA

STATE OF OKLAHOMA, ex. rel. W.A. DREW EDMONDSON, in his capacity as ATTORNEY GENERAL OF THE STATE OF OKLAHOMA and OKLAHOMA SECRETARY OF THE ENVIRONMENT, J.D. Strong, in his capacity as the TRUSTEE FOR NATURAL RESOURCSE FOR THE STATE OF OKLAHOMA,	)	
Plaintiffs,	)	
	)	**Case No. 05-CV-329-GKF-SAJ**
v.	)	
TYSON FOODS, INC., TYSON POULTRY, INC., TYSON CHICKEN, INC., COBB-VANTRESS, INC., AVIAGEN, INC., CAL-MAINE FOODS, INC., CAL-MAINE FARMS, INC., CARGILL, INC., CARGILL TURKEY PRODUCTION, LLC, GEORGE'S INC., GEORGE'S FARMS, INC., PETERSON FARMS, INC., SIMMONS FOODS INC., and WILLOW BROOK FOODS, INC.,	)	
Defendants.	)	

EXPERT REPORT OF

John P. Connolly, Ph.D., P.E., B.C.E.E.

Quantitative Environmental Analysis, LLC

January 30, 2009

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List of Acronyms

ADEQ	Arkansas Department of Environmental Quality
Al	Aluminum
ANOVA	Analysis of Variance
As	Arsenic
AWRC	Arkansas Water Resources Center
BN	Bioavailable Nitrogen
BP	Bioavailable Phosphorus
BUMP	Beneficial Use Monitoring Program
Ca	Calcium
CDM	Camp, Dresser, and McKee
CFR	Code of Federal Regulation
CIC	Constant Initial Concentration
Cs	‘Cesium
Cu	Copper
DAP	Diammonium Phosphate
DOM	Dissolved Organic Matter
DQO	Data Quality Objective
DRP	Dissolved Reactive Phosphorus

EOF	Edge of Field
Fe	Iron
GIS	Geographical Information System
IBI	Index of Biotic Integrity
IRW	Illinois River Watershed
K	Potassium
LAL	Litter Applied Land
Mg	Magnesium
Na	Sodium
NAWQA	National Water-Quality Assessment Program
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
NTU	Nephelometric Turbidity Unit
ODEQ	Oklahoma Department of Environmental Quality
ODWC	Oklahoma Department of Wildlife Conservation
OM	Organic Matter
OSU	Oklahoma State University
OWRB	Oklahoma Water Resources Board
P	Phosphorus
PBSJ	Post, Buckley, Schuh, and Jernigan
PCA	Principle Components Analysis
PSD	Proportional Stock Density
QA/QC	Quality Assurance/Quality Control
QAPP	Quality Assurance Project Plan
QEA	Quantitative Environment Analysis, LLC
qPCR	Quantitative Polymerase Chain Reaction
RSD	Relative Stock Density
SOP	Standard Operating Procedure
SPLP	Synthetic Precipitation Leachate Procedure
SRP	Soluble Reactive Phosphorus
STP	Soil Test Phosphorus

TDS	Total Dissolved Solids
TMDL	Total Maximum Daily Load
TSI	Trophic State Index
TSP	Triple Superphosphate
TSS	Total Suspended Solids
TWCA	Texas Water Conservation Association
USACE	United States Army Corps of Engineers
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WASP	Water Quality Analysis Simulation Program
WEP	Water Extractable Phosphorus
WQT	Water Quality Criteria Threshold
WWTP	Wastewater Treatment Plant
Zn	Zinc

SECTION 1
DECLARATION AND SUMMARY OF FINDINGS

1.1 DECLARATION

My name is John P. Connolly, and I am the President and Senior Managing Engineer for Quantitative Environmental Analysis, LLC, located in Montvale, New Jersey. I hold a BE degree in Civil Engineering from Manhattan College, a ME in Environmental Engineering from Manhattan College, and a PhD in Environmental Health Engineering from The University of Texas at Austin. I am a registered professional engineer in New York and Texas, a Diplomate by Eminence in the American Academy of Environmental Engineers and a member of the United States Environmental Protection Agency (USEPA) Science Advisory Board. For the past 30 years I have studied the fate and impact of pollutants in surface waters as a research engineer at Manhattan College, a research scientist with the USEPA, a professor of Environmental Engineering at Manhattan College and a consulting engineer. My work has focused on understanding and predicting the relationships between pollutant discharges and water quality. I have worked on more than 50 water quality projects and on several major projects related to the issues of algal blooms and dissolved oxygen depletion. Among these are studies of eutrophication in Lake Erie, the Delaware Estuary, Mirror Lake, NH, the Androscoggin River, ME and a series of reservoirs along the Lower Colorado River in Texas. I have been involved in government funded research to advance the state of the art of eutrophication modeling and contributed to the development of the USEPA supported Water Quality Analysis and Simulation Program (WASP) model commonly used to model algae growth in reservoirs. Many of my projects have involved developing an understanding of the contributions of multiple potential sources to observed water quality problems. My full resume can be found in Appendix A.

I was asked to evaluate whether the use of poultry litter as a fertilizer in the Illinois River Watershed is causing water quality problems in the Illinois River and Lake Tenkiller.

My investigation consisted of assessing the water quality of the Illinois River and Lake Tenkiller and investigating the factors controlling that quality. I focused on nutrient (specifically phosphorus) and bacterial pollution as these are the primary focus of allegations made by the plaintiffs in this case. I also considered other chemicals that were used by the Plaintiffs' consultants as tracers of pollutant sources. Quantitative Environmental Analysis, LLC was compensated at a rate of $348 per hour for the time I devoted to this project.

1.2 SUMMARY OF FINDINGS

1. Poultry litter is not the major source of phosphorus to the Illinois River in Oklahoma.

2. Phosphorus has minimal impact on the water quality of the Illinois River in Oklahoma.

3. Phosphorus impacts only a small portion of Lake Tenkiller.

4. Bacteria sources cause little risk of gastrointestinal illness for recreational users of the Illinois River in Oklahoma.

5. The water quality in the Illinois River Watershed is comparable to other waters in Oklahoma.

6. Water quality is improving in the Illinois River and Lake Tenkiller.

7. The water quality modeling conducted by the Plaintiffs' consultants is flawed and provides no means to assess phosphorus impacts.

8. The ambiguity of the standard operating procedures written for the field staff, as well as the lack of field documentation, calls into question the quality of some of the data collected by the Plaintiff.

SECTION 2
POULTRY LITTER IS NOT A MAJOR SOURCE OF PHOSPHORUS TO THE ILLINOIS RIVER IN OKLAHOMA

2.1 SUMMARY OF DETAILED FINDINGS

- Naturally occurring phosphorus compounds, particularly dissolved inorganic phosphates available for algal growth, are the only forms of phosphorus at issue in this case.
- There are many contributors of phosphorus to the Illinois River and Lake Tenkiller.
- The pollutant fingerprints in the Illinois River and Lake Tenkiller do not match that of poultry litter.
- Poultry litter does not produce more phosphorus runoff than cattle manure or any other fertilizer applied intentionally or naturally (grazing cattle).
- Poultry litter does not produce more phosphorus than other applied fertilizer.
- Changes in water quality in Lake Tenkiller do not track with changes in poultry production.
- The similarity of water quality in Lake Tenkiller and other lakes in the region indicates that the use of poultry litter in the Illinois River Watershed does not degrade water quality beyond what occurs because of development for agriculture and urbanization and the nature of run-of-river reservoirs.
- Wastewater treatment plants appear to be the most important source of bioavailable phosphorus to the system.
- Lake sediment phosphorus is a minor source of bioavailable phosphorus.

2.2 NATURALLY OCCURRING PHOSPHORUS COMPOUNDS, PARTICULARLY DISSOLVED INORGANIC PHOSPHATES AVAILABLE FOR ALGAL GROWTH, ARE THE ONLY FORMS OF PHOSPHORUS AT ISSUE IN THIS CASE

Phosphorus is found in the environment as part of various inorganic and organic substances.[1] The inorganic substances include mineral phosphates and phosphate ions. The organic substances are biological residues. In the aquatic environment (i.e., lakes and rivers), some phosphorus-containing substances are dissolved in the water, others are particles. The particles include mineral phosphates, organic particles and inorganic particles to which phosphates have attached (Figure 2-1). Some forms of organic and inorganic phosphorus dissolve in the water. Dissolved organic phosphorus can be mineralized by bacteria to form dissolved inorganic phosphorus. This form of phosphorus, which is also called soluble reactive phosphorus, is consumed by algae for growth.

Dissolved inorganic phosphorus in lakes and rivers originates from several places: mineralization of dissolved organic phosphorus; dissolution of particulate inorganic phosphorus; or direct input from an outside source such as a wastewater treatment plant. Phosphorus is a micronutrient necessary for the growth of algae that form the base of the river and lake food webs. However, too much dissolved inorganic phosphorus under certain conditions may result in enough algae growth to cause aesthetic and dissolved oxygen problems.[2]

[1] Because phosphorus readily binds with other elements, phosphorus does not exist in its elemental form in the environment.

[2] Dr. Olsen (2008) in Section 6.4.3.5 of his report lists numerous "hazardous" substances allegedly found in poultry litter. Dr. Coale (2008) indicates that a number of the chemicals listed in Olsen (2008) are not commonly found in poultry litter (see Coale 2008; Opinion 3l.) Also, the phosphorus referred to in Table 302.4 of 40 CFR § 302.4 is elemental phosphorus, which is does not naturally occur in the environment. Consequently, of the chemicals remaining in Dr. Olsen's list, only one, ammonia, can contribute to eutrophication. It's generally accepted that phosphorus is the nutrient controlling eutrophication in the system; therefore, the focus of my report is on phosphorus. Ammonia and some of the other chemicals listed by Dr. Olsen can cause fish toxicity; however, none of the State's assessments of the waters within the Illinois River Watershed indicate any issues with fish toxicity (see Section 6 for further discussion).

2.3 THERE ARE MANY CONTRIBUTORS OF PHOSPHORUS TO THE ILLINOIS RIVER AND LAKE TENKILLER

Drs. Cooke, Welch, Fisher, and Engel argue that phosphorus from poultry litter has degraded water quality in the Illinois River Watershed (Cooke and Welch 2008; Fisher 2008; Engel 2008). In doing so, they overstate the degree of water quality degradation and fail to properly account for the other sources of phosphorus in the watershed. Both point and non-point sources exist, as illustrated in the conceptual diagram in Figure 2-2. Livestock and application of commercial fertilizers are significant (see Sections 2.5 and 2.6), as are the sources that result from population growth, particularly the deforestation and urbanization associated with such growth (Grip 2008).

Deforestation is a well-documented cause for increased non-point source loadings to receiving waters. Forests tend to be relatively good "conservers" of sediment and nutrient loads. But, when forests are cut down for logging, development, or other purposes, the soils are less "protected" from erosion events and large sediment loads from deforested lands can occur. These sediment loads carry with them numerous constituents, including phosphorus. When cleared forest lands are developed, further increases in non-point source loading occur because development introduces impervious cover (e.g., parking lots, roads, rooftops, etc.). Schueler (2000) found that the first pulse of non-point source loadings due to urbanization occurred during the construction phases. Schueler (2000) also indicated that the influence of impervious cover carries throughout the lifetime of the urban environment. In fact, Schueler (2000) cites a second and possibly larger sediment pulse from streambank erosion, which is a result of increased storm peak flows and water volumes that occur when a watershed becomes urbanized.[3] Urbanization also causes pollution from point source discharges of wastewater treatment plants (Brinkmann 1985).

Because urbanization degrades water quality, measures of urbanization are typically used to evaluate the potential impacts of development on water quality. Randhir (2003) indicated

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[3] Haraughty (1999) cites bank erosion along the Illinois River and its tributaries as a "substantial threat" to the Illinois River Watershed and indicates this erosion is likely due to degraded riparian areas, roads, and bridges. Grip (2009) also indicates that streambank erosion has occurred in the watershed.

"…impervious cover is widely used as an indicator of water resources degradation." Horner et al. (1997) and Lindsey et al. (1997) concluded that impervious cover affects water quality and quantity and disrupts the natural ecosystem. Research has shown that even small percentages of urbanization impact water quality. The Randhir (2003) study found that sediment loadings were "especially high" from sub-basins with urbanization as low as 3%. Booth and Reinelt (1993), Horner et al. (1997), and May et al. (1997) all found that stream channels experience consistent bank erosion during storm events even when their contributing watersheds have relatively low impervious cover.

Besides non-point source pollution, the population increase in the Illinois River Watershed has increased pollution via septic systems (Sullivan 2009) and increased wastewater treatment plant discharges (Jarman 2008). Because much of the phosphorus load from a wastewater treatment plant is dissolved and bioavailable for growth of algae, the increased discharge from wastewater treatment plants can sometimes be more of a concern than the non-point source pollution caused by urbanization. Even if centralized wastewater treatment is not employed to deal with increasing populations, the increased number of septic systems can potentially contaminate groundwater (and subsequently, surface water).

Activities on the Illinois River Watershed other than poultry litter application impact water quality. Even small percentages of impervious cover (percentages that are seen in the Illinois River Watershed) can cause water quality degradation. Consequently, these activities cannot be ignored.

2.4 THE POLLUTANT "FINGERPRINTS" IN THE ILLINOIS RIVER AND LAKE TENKILLER DO NOT MATCH THAT OF POULTRY LITTER

According to the Plaintiffs' consultants Drs. Fisher and Olsen, poultry litter is the primary source of the phosphorus in Lake Tenkiller and the streams within the Illinois River Watershed. They base this argument on their belief that poultry litter has a unique chemical signature that is maintained as runoff carries some of the chemicals in poultry litter from litter-applied land (LAL) through the watershed to the Illinois River and through the river to Lake

Tenkiller. However, they make no effort to demonstrate how chemicals in poultry litter would make their way across the land to local streams, down these streams to the Illinois River and through the Illinois River to Lake Tenkiller, nor what would happen to these chemicals along this journey. Such a journey is a complex adventure impacted by complex physics, chemistry, and biology. Along the way, chemicals are diluted, trapped, and transformed. Typically, complex fate and transport models are developed to account for all the processes at work. Drs. Fisher and Olsen ignored all of this. Dr. Engel made an attempt at modeling, but he too ignored many of the important processes and Dr. Bierman, in his January 2009 Expert Report on behalf of the Defendants, documents the failings of Dr. Engel's efforts.

Drs. Fisher and Olsen relied on the naive and untested assumption that the relative concentrations of the chemicals in poultry litter are immune to the complex physics, chemistry and biology; or, more simply, whatever happens to one chemical happens to all chemicals. The relative concentration pattern in poultry litter was treated as a chemical signature or fingerprint they compared to the chemical signature of water and sediment samples. Dr. Fisher relied primarily on four chemicals as the basis for this signature: phosphorus, zinc, copper, and arsenic. Using these chemicals, Dr. Fisher concludes that sediments of the Illinois River and Lake Tenkiller are a mixture of uncontaminated soil and poultry litter. Dr. Olsen used these and other chemicals, which he subjected to a Principal Components Analysis (PCA), in an effort to ascribe the source of the chemicals found at various points in the river. Relying principally on water column sample data, he concludes that poultry litter has impacted most locations sampled in the watershed.

I have examined the Illinois River Watershed water quality data in various ways to determine whether Drs. Fisher and Olsen are correct; i.e., there is a chemical signature in environmental samples that matches the chemical signature of poultry litter. I began by comparing the relative concentrations of the major cations and the major metals in various environmental media.

Figures 2-3 and 2-4a and b, respectively show the relative concentrations of the major cations calcium (Ca), sodium (Na), potassium (K) and magnesium (Mg; used by Dr. Olsen) and the metals zinc (Zn), copper (Cu), and arsenic (As; used by Drs. Olsen and Fisher) for water purported to represent runoff from fields to which poultry litter had been applied and river and stream samples under base flow and high flow and Lake Tenkiller. Calcium and magnesium cause water hardness and are present in similar proportions in all types of samples, reflecting their abundance throughout the watershed at concentrations that yield water hardness in the range of 120 to 154 mg/L as calcium carbonate. (This hardness range is characteristic of streams in eastern Oklahoma; http://water.usgs.gov/owq/hot.html). Lake Tenkiller exhibits somewhat lower calcium concentrations, possibly due to precipitation of calcium carbonate during the summer. Overall, calcium and magnesium concentrations are driven by the mineral composition of the watershed soils and provide no means to track poultry litter or other sources. Looking at potassium and sodium, the edge-of-field (EOF) samples from alleged poultry litter amended fields are unique; they have on average about two times more potassium than sodium. The stream and lake samples show the opposite pattern; they have more sodium than potassium in a proportion that is similar among base flow, high flow, and lake samples. Thus, the cation data provide no evidence that poultry litter has impacted streams in the watershed or Lake Tenkiller.

Looking at the major metals, the dominant metal in poultry EOF samples is Cu, whereas it is Zn in all the river, stream, and lake samples. Copper accounts for 54% of the sum of these metals in EOF samples, but less than 14% in river, stream, and lake samples. This poor match exists in soil and sediment samples as well, as shown in Figure 2-5. Thus, the major metals data provide no evidence that streams in the watershed or Lake Tenkiller have been impacted by poultry litter. This conclusion is supported by more detailed examination of the data that I will now describe.

The mean ratio of total Zn to total Cu for samples of poultry litter, alleged poultry litter amended soils, EOF samples from alleged poultry litter amended fields, river water and sediments, and Lake Tenkiller water and sediments are shown in Figure 2-6. An analysis of variance (ANOVA; $F_{6,540} = 69.5$, $P < 0.0001$; Tukey HSD $\alpha < 0.05$) indicates statistical differences in the mean total Zn to total Cu ratio among some of these sample groups (compare letters in Figure 2-6, different letters indicate statistically different ratios). The mean total Zn to total Cu ratios in poultry litter and alleged poultry litter amended soil are 1.5 and 2.1

respectively, and these ratios are not statistically different from each other. The ratio in the poultry EOF samples is 3.5, statistically higher than poultry litter. The increasing concentration of Zn relative to Cu moving from the litter to the soil and to EOF water, and the statistical difference between litter and EOF samples suggests that there is an additional source of Zn in the soil and/or that the transport rate of Zn exceeds that of Cu. Illinois River Watershed river water and sediment, Lake Tenkiller water and sediment, and Illinois River Watershed soils not exposed to poultry litter (control soils) have total Zn to total Cu ratios between 5.0 to 8.9, all substantially higher than the poultry associated sample groups (Figure 2-6) and statistically different from the poultry groups.[4]

Further insight regarding the differences in metal composition between poultry litter and EOF can be gained by looking at individual samples. As shown in Figure 2-7, the water at the edge of litter-applied fields (triangular symbols) can have concentration ratios very different from litter (diamond symbols). The plots in this figure show the samples arranged in order of increasing concentration ratio and plotted according to the probability of having a lesser ratio. For example, half the values have a lesser ratio than the value plotted at 50%. The water running off the litter-applied field can have as little as 6 times as much phosphorus as Zn (i.e., a concentration ratio of 6) or as much as 200 times as much phosphorus as Zn. In contrast, excepting two outliers, the poultry litter samples all have ratios between 30 and 60. About 60% of the EOF samples have ratios in particulate matter below 30, suggesting that the runoff tend to have less phosphorus per unit Zn than the litter. The opposite is true for Cu; these eroded particles have up to 700 times more phosphorus than Cu, with about 60% of the samples having a phosphorus-to-copper ratio greater than the maximum measured in litter of 60.

There are also differences in the how Cu and Zn are transported to the EOF; much more of the Cu is dissolved (a median close to 90% versus a median of about 50% for Zn). This concurs with the fact that Cu tends to be bound to dissolved organic matter (DOM).

[4] Total zinc and copper levels in Lake Tenkiller water were usually below detection limit (only 6 detects in 74 samples), therefore Lake Tenkiller water is excluded from the comparative statistical analysis.

The differences in chemical signature between poultry litter and EOF samples show that the efforts of Drs. Olsen and Fisher were doomed from the start. Their fundamental assumption of a largely invariant chemical signature is false. There is no unique chemical signature for poultry litter that can be traced through the watershed. Moreover, many potential sources of phosphorus contain the chemicals Drs. Olsen and Fisher used to track sources and most of these chemicals have unique behavior in the environment. Consider phosphorus, zinc, copper, and arsenic; the focus of Dr. Fisher and an important part of the focus of Dr. Engel. Each of these chemicals undergoes unique chemical reactions that govern the extent to which they can be taken up by plants, travel with flowing water, form insoluble compounds, and become attached to particulate matter. The concentration of metals in runoff depends on the soil properties (i.e., pH, clay content, presence of DOM) and application history (Arias et al. 2005). In general terms, arsenic is more mobile than Zn and Cu (Gupta and Charles 1999). Copper has a higher affinity for organic matter than Zn (Impellitteri et al. 2002). In particular, Cu is documented to preferentially bind to DOM (Han and Thompson 2003; Romkens et al. 2004; Arias et al. 2005; Lu and Allen 2001; Grassi et al. 2000; Karathanasis 1999). Therefore, Cu mobility is highly dependant on the abundance and composition of organic matter in the litter and soil and the extent to which organic matter is leached from the soil during runoff events. In a two-year study using lysimeters in Arkansas, Pirani et al. (2006) found that only 0.3% of the Cu added to the soil by poultry litter application leached, whereas 49% of the applied Zn leached. A similar result is suggested by the soil sampling performed by Gupta and Charles (1999). Arias et al. (2005) showed that absorption/desorption hysteresis is higher for Cu than for Zn, adding to the reduced mobility of Cu as compared to Zn. On the contrary, Keller et al. (2002) found completely different behavior for sewage sludge applied to soil. In this case, the concentration of Cu in runoff was higher than the concentration of Zn. The Cu was mainly found in dissolved form bound to DOM, which leached from the sludge. These studies show that the mobility of Cu and Zn are not the same and are also not directly proportional to the soil concentration in all cases.

In addition to the differences in the availability of the different compounds, their natural abundance differs in different soils. For example, Zn is present in control soils at concentrations similar to concentrations found in litter-amended soils (Gupta and Charles 1999;

Pirani et al. 2006). The average values for total Zn concentration in alleged LAL samples is 34 mg/kg, which is in the same order as the value reported by Gupta and Charles (1999) for control soils (20 to 34 mg/kg).

The site-specific data as well as the published studies demonstrate the futility of using ratios among phosphorus, zinc, and copper to trace the fate of phosphorus applied to fields as part of poultry litter. There is no unique ratio among phosphorus, zinc and copper that can be used to determine how much, if any, of the phosphorus in lake or stream sediments came from poultry litter. Unfortunately, Dr. Fisher, ignoring this fact, relied on gross data comparisons that obfuscate important differences and inconsistencies that exist when samples of litter, EOF water, and sediments are compared.

Dr. Fisher presents graphs that purport to show that the stream and lake sediments are a simple mixture of poultry litter and control soils (Figures 24 and 32 of Fisher [2008]). A close examination of these figures reveals the problems just discussed. The data exhibit tremendous variability and the sediment data fail to fall along the lines that supposedly reflect the mixing of poultry litter and control soils. In fact, the differences between the data and the mixing line, which are hard to see because of the scales of the graphs, are substantial. Relying on average concentrations, those differences are illustrated below.

Mixing control soil and poultry litter would generate the phosphorus, zinc, and copper concentrations shown as lines on the graphs in Figure 2-8. The lines start at the lower left at the composition of control soil and move up and to the right as the amount of poultry litter increases. The graphs also show the phosphorus, zinc, and copper concentrations measured in lake sediments. None of the lake sediment samples match the control soil-poultry litter mixture lines. In other words, the sediment is not a mixture of control soil and poultry litter. The source identification method used by Dr. Fisher provides no useful information about the contribution of poultry litter or any other pollutant source to lake or stream sediment.

2.5 POULTRY LITTER DOES NOT PRODUCE MORE PHOSPHORUS RUNOFF THAN CATTLE MANURE OR ANY OTHER FERTILIZER APPLIED INTENTIONALLY OR NATURALLY (GRAZING CATTLE)

According to Drs. Olsen and Fisher, runoff from poultry litter amended fields is more contaminated than runoff from fields with grazing cattle. To justify this assertion, Dr. Olsen used EOF samples and the results of a synthetic precipitation leachate procedure (SPLP). The conclusions derived from the analysis of both datasets are wrong as I explain below.

The EOF samples were collected during or shortly after a storm event at more than 80 sites that purportedly drained fields to which poultry litter had been applied. Dr. Olsen compared the mean values of these samples to the mean of two EOF samples taken from sites with grazing cattle. In addition to relying on a specious comparison of incompatible sample sizes (i.e., 80 to 2), he did not determine whether the difference in mean values was statistically significant (i.e., that it signified a real difference).

A thorough analysis shows that for many of the contaminants of interest, the EOF concentrations for cattle and poultry are not statistically different. Most importantly, there is no difference for phosphorus. Figure 2-9 presents a probability plot of the concentration of total phosphorus found in the EOF samples for poultry litter and grazing cattle fields. As can be seen in the plot, the phosphorus concentrations measured in poultry litter fields are comparable to concentrations measured in grazing cattle fields. It is impossible to conclude from these data that the poultry litter fields contribute more phosphorus to runoff than do grazing cattle fields.

Dr. Olsen uses the SPLP study to compare the "leaching potential" of poultry litter and cattle manure. However, this study is not relevant to what happens when it rains on fields containing poultry litter or cattle manure because it was conducted on samples poultry litter or cattle manure and not on amended soil samples. Once applied to the soil, poultry litter and cattle manure change in ways that modify the availability of phosphorus and trace metals. For that reason, the SPLP study on the litter and manure samples does not predict concentrations of phosphorus and metals in runoff. Dr. Olsen acknowledged this fact; in his report he states that the results from the SPLP study are maximum quantities and that, "as shown by Dr. Engel's calculations, the actual quantities leached in the environment are substantially less". Dr. Olsen works around this fact by contending that the relative leaching potential is predictive of the relative concentrations likely to be found in runoff. This argument is wrong because the extent of leaching depends on how the environment affects the composition and characteristics of the litter and manure.

To validate his analysis, Dr. Olsen cites conclusions from a paper by Sauer et al. (1999) comparing runoff from plots amended with poultry litter and cattle manure and subjected to synthetic rain. Dr. Olsen highlights the conclusion that the poultry amended plots "provided at least six times the amount of each nutrient" than the cattle amended plots. However, the paper also states the following:

> *...since the amount of nutrients transported was proportional to the amount applied, losses from the dairy manure and urine treatment were influenced by the assumptions used in determining the amount of feces and urine to apply. Clearly, grazing intensity and waste deposition patterns create potential for large degree of spatial and temporal variability of nutrient runoff from grazed pastures. Further studies in this area are warranted, especially as the potential for nutrient runoff from applied poultry litter diminishes with time after application, whereas, grazing animals continue to deposit wastes on soil surface throughout the growing season.*

In other words, the authors acknowledge that the study is not conclusive of the effect of poultry litter and cattle manure in a field situation due to differences in the applied amount of cattle manure and the effect of aging. Additionally, the study did not consider the effect of the cattle manure deposition patterns (i.e., in general in shaded areas and close to the water [Schomberg et al. 2000; Wells and Dougherty 1997]) and the reduction in soil permeability caused by the treading of the cattle (Wells and Dougherty 1997).

Most importantly, Olsen's analysis of the Sauer study is incomplete because he did not include the fact that the study clearly shows the important effect that aging (soil-litter and soil-manure interaction over time) can have in the runoff from different amendments. Sauer's study comprises two synthetic precipitation events, the first one day after the application of the wastes to the plots and the second two weeks later. During the first event, the nutrient runoff concentrations from fields amended with poultry litter was higher than from the cattle manure plot. However, during the second event the nutrient runoff concentrations from the poultry litter-amended and cattle-grazed fields were not significantly different, as shown by a statistical analysis in Sauer's paper. This result clearly demonstrates the effect that aging can have in only 14 days. The effect of aging on phosphorus runoff was specifically noted by Sharpley (1997) and Kleinman and Sharpley (2003). In particular, Sharpley studied 10 different poultry litter applied soils in Oklahoma. He found that the time elapsed between litter application and rainfall significantly affects the runoff concentration of nutrients and recommended avoiding litter application during periods of high rainfall probability. This recommendation is one of the current United States Department of Agriculture (USDA) best practices for poultry litter application (Sharpley 2006).

Dr. Olsen does not consider the wealth of published evidence that runoff water quality depends on many variables and no general conclusion can be drawn signaling one fertilizer as inherently worse than others. In fact, the concentration of chemicals in runoff water depends not only on the concentration of those chemicals in the applied manure, but also on manure properties, manure application methodology, soil properties of the field where the manure is applied, runoff hydrology of the area, management practices related to erosion control, animal access to water bodies, and the chemical element being considered. For example, Sharpley (2006) stated that up to 80% of the total phosphorus in runoff water can be controlled through best management practices. Based on available literature, the following paragraphs present an analysis of the relative impact of poultry litter and cattle manure to water quality in the Illinois River Watershed.

There have been extensive efforts to relate phosphorous concentration in soil and the dissolved reactive phosphorus (DRP) concentration in runoff water. Pote et al. (1999) found that runoff did not always correlate with soil test phosphorus (STP) but was well correlated with the water extractable phosphorus (WEP) content of the soils. Subsequently, Kleinman et al. (2002a) and DeLaune et al. (2004) showed that the phosphorus concentration in runoff from a field amended with manure was correlated with the WEP of the applied product. The concentration of total phosphorus in poultry litter is typically about 4 times higher than in cattle manure. However, Bremer et al. (2008) and Kleinman et al. (2005) show that the WEP is in the same range. Kleinman et al. (2005) reported average values of WEP of 2.3, 3.2, and 4.0 g/kg for beef cattle, poultry (broilers), and dairy manures, respectively. Dr. Olsen, in Table 6-4-1 of his report, confirms this similarity, showing an average WEP value for 16 samples of poultry litter of 1.44 g/kg and values of 3.02 and 0.95 for 5 samples of fresh and dry cattle manure, respectively. Therefore, if all other conditions are the same, similar phosphorus runoff is expected from soils amended with either litter or manure.

Mass balances of WEP can be calculated for the Illinois River Watershed using the published results from Kleinman et al. (2005), Dr. Olsen's report (2008), and the manure production values from Dr. Clay's Expert Report (2008). Tables 2-1 and 2-2 summarize the mass balances and show that non-poultry livestock produce more WEP than poultry livestock. The mass balance constructed using Dr. Olsen's data (Table 2-1) shows that non-poultry sources contribute 68% more than poultry livestock. The difference is smaller using Kleinman's data (Table 2-2) but still shows that non-poultry sources contribute 14% more WEP than poultry sources. Both mass balances indicate a bigger input of WEP from non-poultry sources even though they significantly under-represent non-poultry sources because they do not include horses, sheep, and wildlife.

Table 2-1. Water extractable phosphorus mass balance in the Illinois River Watershed.

Livestock	Average Dry Matter (%)	Average WEP g/kg (dry basis)	Average WEP g/kg (as deposited)	Annual Manure Contribution Tons	Annual WEP Contribution Tons
Beef cow	25%	3.0	0.76	1,870,847	1,422
Dairy cow	15%	4.0	0.60	154,296	93
Swine	20%	9.2	1.84	362,331	667
				Total Non-poultry	**2,182**
Layers	58%	4.9	2.84	113,141	322
Broilers	82%	1.4	1.18	691,234	812
Turkey	75%	6.3	4.73	35,397	167
				Total Poultry	**1,301**

Notes:
Beef cow dry matter and WEP (dry basis) from Dr. Olsen report Table 6.4-1. It was assumed that 90% of the manure is dry and 10% is fresh (Olsen 2008, page 6-11).
Broilers dry matter and WEP (dry basis) from Olsen (2008).
All other values for Average Dry Matter and Average WEP (dry basis) were extracted from Kleinman et al. (2005).
Manure contribution data from Clay (2008).

Table 2-2. Mass of Water Extractable Phosphorus (WEP) generated in the Illinois Watershed using Kleinman et al. (2005) data.

Livestock	Average Dry Matter (%)	Average WEP g/kg (dry basis)	Average WEP g/kg (as deposited)	Annual Manure Contribution Tons	Annual WEP Contribution Tons
Beef cow	37%	2.3	0.85	1,870,847	1,592
Dairy cow	15%	4.0	0.60	154,296	93
Swine	20%	9.2	1.84	362,331	667
				Total Non-poultry	**2,351**
Layers	58%	4.9	2.84	113,141	322
Broilers	71%	3.2	2.27	691,234	1,570
Turkey	75%	6.3	4.73	35,397	167
				Total Poultry	**2,059**

Notes:
Average Dry Matter and Average WEP (dry basis) values were extracted from Kleinman et al., 2005.
Manure contribution data from Dr. Clay (2008).

2.6 POULTRY LITTER DOES NOT PRODUCE MORE PHOSPHORUS RUNOFF THAN OTHER APPLIED FERTILIZERS

As mentioned before, there have been many studies describing the phosphorus runoff from different kind of organic and inorganic fertilizers. The following paragraphs summarize some of the relevant studies that show the runoff of phosphorus is not typically higher in poultry litter amended soils than in soils amended with other organic and inorganic fertilizers.

Hall et al. (1994) measured the runoff from fields amended with commercial inorganic fertilizers at the recommended soil test rate and poultry litter at 10 tons per ha (4 tons per acre), which is a typically recommended rate in published agronomic guidelines. They found that the total mass of nitrogen in the runoff was about 25% lower from the poultry litter amended fields than for the commercial fertilizer amended ones. The losses of phosphorus due to runoff were about the same for both fields. It is interesting to note that Dr. Engel calculated a maximum poultry litter application in the area of about 2.3 tons per ha (0.9 tons per acre), which is more than four times lower than the applied in Hall's study and well below the typical recommended values. These facts suggest that the application of poultry litter in the Illinois River Watershed has been below normal standards and that the expected nutrient runoff losses would be lower than using a commercial fertilizer to increase crop production. DeLaune et al. (2004) also obtained much higher phosphorus runoff values using commercial fertilizers (in this case triple superphosphate [TSP]). All the fertilizers in the DeLaune study where applied at the same rate (78 kg P/ha) and the soluble reactive phosphorus (SRP) runoff from TSP was more than 6 times higher than from poultry litter, likely due to the fact that most of the phosphorus in commercial fertilizers is WEP and therefore more available to runoff.

Similar results were obtained by Kleinman et al. (2002b) in a study of the application of fertilizer at a rate of 100 kg/ha of total phosphorus from 4 different sources (poultry manure, dairy manure, swine, and diammonium phosphate [DAP]) on three different soils. They found no statistical difference in runoff DRP concentration from soils subject to DAP, poultry, or swine manure application. Runoff from soils subject to dairy manure was lower for two of the soils and the same for one of the soils illustrating the aforementioned impact of soil properties on runoff of the applied product. It should be noted that the application refers to poultry manure and not poultry litter; poultry litter has a different behavior than manure (i.e., typically litter is dryer and has less WEP than manure [Vadas and Kleinman 2006; Sharpley et al. 2004]).

A 2004 study performed by Sharpley et al. focused on the impact of long-term application of poultry manure, poultry litter, swine slurry and dairy cattle manure on soil phosphorus concentration. This study included fields in Oklahoma, New York, and Pennsylvania that had been subject to continuous application of these materials as fertilizer for more than 10 years. The application of waste to these fields was based on total phosphorus and annually ranged between 75 and 150 kg/ha for poultry litter and dairy manure. Using the average total phosphorus concentration of poultry litter, this application rate corresponds to between 5 to 10 tons per ha of litter (2 to 4 tons per acre), which is within the recommended application range. The soil analyses performed by this study show that the soils amended with dairy manure have higher total phosphorus and WEP than the soils amended with poultry litter. Average WEP for dairy manure amended soils was 40.2 mg/kg, whereas for the poultry litter amended soils the average value was 20.8 mg/kg. As presented before, there is strong correlation between WEP contents of the soil and runoff so these values imply that the dairy manure amended soils analyzed in this study will have higher phosphorus runoff than the soils amended with poultry litter.

From the studies summarized in the previous paragraphs, it can be concluded that the potential runoff from a fertilized field will depend on many variables like soil characteristics, field management, rate, and history of application, etc. Therefore, it is erroneous to generalize that a certain fertilizer generates a bigger impact than others as they all have potentially the same impact if the application meets best management practices.

2.7 CHANGES IN WATER QUALITY IN LAKE TENKILLER DO NOT TRACK WITH CHANGES IN POULTRY PRODUCTION

Dr. Fisher attempted to use the phosphorus concentrations in dated sediment cores from Lake Tenkiller to infer the historical trend in phosphorus loading to the lake. He compared this trend to the trend in poultry house density and concluded that they match. Unfortunately, he made three mistakes that invalidate this comparison:

- he incorrectly dated the sediment cores, thus skewing the phosphorus concentration trends;
- he did not account for variations in sediment phosphorus caused by variations in the iron and aluminum content of the sediment rather than variations in phosphorus loading to the lake; and
- he ignored the fact that phosphorus levels in the lake sediments have not risen since the late 1980s and may be trending downward, indicating that the phosphorus loading to the lake has not risen despite increases in poultry population.

2.7.1 Dating of Sediment Cores

Six cores were collected from Lake Tenkiller in August 2005 and four were selected for geochronological and chemical analysis (a fifth was subjected to chemical analysis, only). The cores were sectioned into 2-cm intervals and analyzed for radionuclides, metals, and nutrients. Radionuclides were measured in an effort to determine the age of each section, i.e., the year or years each section was deposited. The goal was to generate a time history of the concentrations of phosphorous and various metals on sediments depositing in the lake. Soster (2005a, 2005b, 2005c, 2006) dated these cores using unsupported (excess) ^{210}Pb activities (measured indirectly by the analysis of its radioactive decay product ^{210}Pb) in the constant initial concentration (CIC) model. Excess ^{210}Pb was calculated by subtracting ^{214}Bi activity, a surrogate for supported ^{210}Pb. The CIC model assumes that all sediments in the core had the same activity of excess ^{210}Pb at the time of deposition (Cohen 2003). Dates of sediment deposition are calculated by fitting the decrease in ^{210}Pb activity with depth to a radioactive decay model. Cesium (C's)-137 was used as an independent means of dating the sediment. The peak ^{137}Cs activity is associated with sediments deposited around 1963 and the first appearance of ^{137}Cs is presumed to represent about 1954.

The ages and sedimentation rates estimated by Soster (2005a, 2005b, 2005c, 2006) and presented by Dr. Fisher, which were calculated from the ^{210}Pb data, put the peak ^{137}Cs activity in sediment deposited much later than 1963. The differences are substantial, as shown in Table 2-3.

Table 2-3. The ^{210}Pb age as calculated by Dr. Fisher for the ^{137}Cs peak (presumed to be 1963).

Core	Year of Deposition of Sediments with the Peak ^{137}Cs Activity, as Assigned by Dr. Fisher	Rounded Difference Between Dr. Fisher's Date and 1963, the Expected Year When Depositing Sediments Would Have Maximum ^{137}Cs
LKSED-01	1970	7 years
LKSED-02	1971	8 years
LKSED-03	1977	14 years
LKSED-04	1982	19 years

Dr. Fisher argues that "concordant ages cannot be obtained for ^{210}Pb and ^{137}Cs methods" due to their differing input signals. This is not correct. The dating derived from ^{210}Pb and ^{137}Cs should be roughly consistent. The two dating methods are almost always used in combination. The ^{137}Cs profiles look good. They have defined ^{137}Cs peaks (~1963) near the bottom of the cores, which makes sense based on the fact that sedimentation would have begun in 1954 after the dam was completed. There is no reason to discount the ^{137}Cs data and therefore the significant differences in the dates determined using the two methods raise doubts about the ^{210}Pb dating analysis.

As seen in Figures 2-10a and 2-10b, two things stand out in the excess ^{210}Pb activity profiles for the Lake Tenkiller cores: 1) the ^{210}Pb activity in the top 6 to 10 cm of the cores is variable, but does not consistently decline with depth, likely due to biological mixing; and 2) the ^{210}Pb activity is relatively constant over the bottom 6 to 10 cm of the cores profile, suggesting rapid deposition of sediments when the lake was first created (i.e., these bottom sediments were all deposited at about the same time). Consequently, the top and bottom portions of the core do not conform to the assumptions of the CIC dating model and should not be used in determining the rate of sedimentation that is the basis for dating the core sections. The analysis was redone excluding the surface and bottom samples indicated by red circles around the data points in Figures 2-10a and b. The log-linear regression of the data from which the dating was determined (i.e., with the surface and bottom sample data points removed) is shown in Figure 2-11. The ages calculated by using only the excess ^{210}Pb activities for the mid-portions of each core coincide closely with the ^{137}Cs age for cores LKSED-02, -03, and -04 (Table 2-4). A discrepancy remains for core LKSED-01. For this core the 38-42 cm sample was pre-dam closure Illinois River Floodplain sediment and provides a reference point of 1954, roughly when the lake began to fill in. The exact depth of the ^{137}Cs peak in this core is uncertain and falls between 30-36 cm. If we assign 1954 to the 40 cm depth in the core, the range of dates for the 30-36 cm section are 1960-1965, coinciding with the ^{137}Cs peak of 1963. Dating this core using ^{210}Pb activity results in much younger ages for the bottom sediment, therefore I used the ^{137}Cs dating results.

Table 2-4. The ^{210}Pb age as calculated by QEA for the ^{137}Cs peak (presumed to be 1963).

Core	Year of Deposition of Sediments with the Peak ^{137}Cs Activity, as Assigned by QEA	Rounded Difference Between QEA Date and 1963, the Expected Year When Depositing Sediments Would Have Maximum ^{137}Cs
LKSED-01	1965	0 years*
LKSED-02	1964	1 year
LKSED-03	1964	1 year
LKSED-04	1974	11 years

**This core was dated based on ^{137}Cs*

2.7.2 Proper Normalization of Sediment Phosphorus Concentrations

Sediments are composed of particulate organic matter, mineral particles, and precipitates. Phosphorus can be present as part of the particles themselves or sorbed to the particles. Particles with a greater amount of iron, aluminum, and calcium tend to have a greater amount of phosphorus. This is because some of the compounds that make up the sediment are combinations of phosphorus and these elements and because phosphorus readily adsorbs to iron and aluminum oxides and hydroxides that are common components of the sediment (Shukla et al. 1971; Detenbeck and Brezonik 1991; Novak and Watts 2006).[5] As a result, the phosphorus concentration of a sediment sample depends on the nature of the sediment. This means that differences in phosphorus concentration alone cannot be used to infer differences in phosphorus loading.

The influence of iron and aluminum on phosphorus concentration is illustrated by examining a few representative stream sediment and control pasture soil samples collected by the

[5] Phosphorus concentration is determined to a lesser extent by other constituents of the sediment, typically the most important of which is organic matter.

Plaintiff's consultants in 2005 and 2006. Table 2-5 and Figure 2-12 show for these few samples the concentrations of total phosphorus, Iron (Fe), aluminum (Al), and the ratio of total phosphorus to the sum of Fe and Al. The total phosphorus concentrations range from 123 to 775 mg/kg, about a factor of six; close to the range of total phosphorus concentrations found in stream sediments (111 to 921 mg/kg by method SW6020B). Note that the samples also have a wide range of Fe + Al concentrations that vary by almost a factor of five and that the total phosphorus concentrations correlate with the Fe + Al concentrations. To account for this correlation, the total phosphorus concentrations were normalized by the sum of Fe and Al (Daskalakis and O'Connor 1995). The normalized concentrations are all very similar, ranging over a factor of two, despite the fact that the total phosphorus concentrations range over a factor of six. Most significantly, the normalized concentrations for the stream samples fall within the range of values for the three control soil samples. In fact, of the many stream samples taken by the Plaintiffs' consultants, only one has a normalized total phosphorus concentration substantively above what was found for the control pasture soils; Station SD-008 (which is not shown in Table 2-5 or Figure 2-12) had a value of 0.044. The striking conclusion from this illustration is that higher concentrations of total phosphorus are not presumptive evidence of an anthropogenic source. These higher concentrations may simply reflect the greater presence of iron and aluminum compounds (or calcium compounds) that naturally contain phosphorus or have the ability to bind phosphorus naturally present in the environment. In other words, there is no evidence that the total phosphorus concentrations in stream sediments are higher than expected from soils running off of control fields. The total phosphorus concentrations differences among the stream sediment samples are the result of differences in the concentrations of Fe and Al in the material settling to the bottom.

Table 2-5. Phosphorus, iron, and aluminum content of a few representative stream and control soil samples.

Sample	Type	Location	Fe (mg/kg)	Al (mg/kg)	Fe + Al (mg/kg)	TP (mg/kg)	TP/ (Fe + Al)	Measurement Basis
SD-029	Sediment	Evansville Creek	4,668	1,383	6,051	123	0.02	Wet Weight
SD-203[1]	Sediment	Evansville Creek	7,904	2,916	10,820	275	0.025	Wet Weight
CL2-B-4	Soil	Nickel Preserve	11,500	6,850	18,350	409	0.022	Unknown
CL2-B-2	Soil	Nickel Preserve	10,800	6,470	17,270	475	0.028	Unknown
CL2-A-2	Soil	Nickel Preserve	12,800	6,690	19,490	518	0.027	Unknown
SD-001[1]	Sediment	Buddy Kidd Creek	19,326	7,869	27,195	775	0.028	Wet Weight

[1]*Average of 2 replicate samples.*

An overall examination of total phosphorus concentration in relation to Fe + Al concentration shows a strong correlation for stream and lake sediments (Figure 2-13). Moreover, these two types of sediment exhibit remarkably similar correlations as evidenced by their plotting on top of each other. Control soil from a field matches this relationship. Poultry litter, which is also shown in Figure 2-13, does not. Strong correlations between phosphorus extracted from sediments and the coextracted iron plus aluminum have been demonstrated in a number of studies (e.g., Danen-Louwerse et al. 1993; Zhou et al. 2005).

The strong association of phosphorus with iron and aluminum compounds is documented for Lake Tenkiller sediment by measurements of the phosphorus associated with readily extractable Fe, Al, and Ca. These analyses were conducted by the Plaintiffs' consultants on 3 segments of 4 sediment cores collected from the riverine (LKSED-4), transitional (LKSED-3) and lacustrine (LKSED-2 and LKSED-1) regions of the lake. The data from these analyses, which are shown in Table 2-6, illustrate the importance of Al and Fe compounds as accumulators of phosphorus and their dominance over Ca compounds (together Al and Fe compounds account for six times more phosphorus than Ca compounds).

Table 2-6. Lake Tenkiller sediment phosphorus that is loosely-bound or associated with extractable Al, Ca, or Fe.

Sample	Al Bound P (mg/kg dry)	Ca Bound P (mg/kg dry)	Fe Bound P (mg/kg dry)	Loosely Bound P (mg/kg dry)
LKSED-4-01-01	124	46.3	232	2
LKSED-4-02-01	199	56.1	140	2
LKSED-4-03-01	246	65.6	121	2.61
LKSED-3-02-01	168	46.5	139	2
LKSED-3-03-01	186	55.2	118	2
LKSED-3-04-01	168	54	166	2
LKSED-2-02-01	164	46.8	126	4.51
LKSED-2-03-01	165	53.3	118	2
LKSED-2-04-01	155	52.3	149	2
LKSED-1-02-01	143	65	230	2
LKSED-1-03-01	168	71.2	310	2
LKSED-1-04-01	174	66.3	314	2.13
Average	**172**	**57**	**180**	**2**

Total phosphorus concentrations in the four lake sediment cores vary with depth as shown in the top row of plots shown in Figure 2-14. Much of this variability disappears when the concentration is normalized by the Fe +Al concentration, as shown in the bottom row of plots in the figure. The normalized concentrations provide a good sense of the trends in total phosphorus loading to the lake over the period over which the sediments accumulated on the lake bottom.

2.7.3 Time Trends in Lake Tenkiller Phosphorus

Using the corrected dating of the segments in the lake sediment cores and the Fe + Al normalized total phosphorus concentrations, the time trend of Lake Tenkiller phosphorus is shown in Figures 2-15 and 2-16. Sediments deposited prior to the mid-1960s had normalized total phosphorus levels of about 0.020 to 0.025 g total phosphorus per g Fe + Al, which are within the range of levels found in field control soils and most stream sediments (i.e., 0.020 to 0.028 g total phosphorus per g Fe + Al). The most upstream core (LKSED-04) shows an increase from the late-60s to the early-80s to about 0.035 g total phosphorus per g Fe + Al followed by a slow decline to about 0.031 g total phosphorus per g Fe + Al in 2005. This upstream core gives the best indication of the trends in total phosphorus load to the lake. The next downstream core (LKSED-03) exhibits less change over time, with concentrations throughout the core remaining in the range seen for field control soils (0.020 to 0.028). The further downstream cores are impacted by phosphorus cycling that occurs due to lake thermal stratification and depletion of oxygen in the hypolimnion. As a result, these cores show a somewhat different time pattern that shows a more gradual increase in concentration to a peak in the late-1980s at about 0.030 to 0.035 g total phosphorus per g Fe + Al and relatively constant concentrations to 2005. These temporal trends concur with Litke (1999) who reports that phosphorous concentrations are decreasing in many National Water-Quality Assessment Program (NAWQA) study units.

The fact that total phosphorus load to the lake reached a maximum in the 1980s and has remained relatively constant or declined slightly since that time is inconsistent with the hypothesis that poultry litter is a dominant source of the total phosphorus in the lake. As shown by Dr. Fisher, the poultry population in the watershed increased substantially over the period between the 1980s and 2005. By his estimate, the population was about 465,000 animal units in 1982, 688,000 in 1992, and 850,000 in 2002 (Smith 2008). Thus, it almost doubled over this 20-year period and increased by about 24% in the last 10 years. Assuming the poultry litter total phosphorus contribution to the lake has been proportional to the poultry population, lake sediment total phosphorus would have risen significantly over the last 20 years if poultry litter was an important total phosphorus source. The absence of a rise means that poultry litter cannot have been an important source.

Another interesting result of the above analysis is that it indicates that in the absence of anthropogenic phosphorus sources the sediments of the river and lake would have about 0.020 to 0.028 g total phosphorus per g Fe + Al. Thus, anthropogenic sources are responsible at most for about 0.01 g total phosphorus per g Fe + Al (i.e., 0.035 minus 0.025), which constitutes about one-third of the phosphorus in the sediments.

2.8 THE SIMILARITY OF WATER QUALITY IN LAKE TENKILLER AND OTHER LAKES IN THE REGION INDICATES THAT THE USE OF POULTRY LITTER IN THE ILLINOIS RIVER WATERSHED DOES NOT DEGRADE WATER QUALITY BEYOND WHAT OCCURS BECAUSE OF DEVELOPMENT FOR AGRICULTURE AND URBANIZATION AND THE NATURE OF RUN-OF-RIVER RESERVOIRS.

The Plaintiffs' consultants contend that poultry litter is the principal cause of water quality problems in the Illinois River Watershed. While they acknowledge the presence of other sources of nutrients and bacteria, they conclude that water quality problems would be minimal in the absence of poultry litter application as a fertilizer. If this conclusion is correct, one expects other reservoirs in nearby watersheds that have much less poultry litter application but similar land use to have better water quality. To test this hypothesis, the water quality of other lakes in the state that have watershed characteristics similar to the Lake Tenkiller watershed, but lower poultry populations, were compared to the water quality of Lake Tenkiller.[6]

Lakes Hugo and Sardis, which are in southeastern Oklahoma (Figure 2-17), were found to be the best available comparisons to Lake Tenkiller. All are in the same general physiographic region of the southern Midwest and contain portions of the Arbuckle and Ozark mountain chains, which are characterized, at least in part, by the presence of karst features including caves and conduits to groundwater recharge, flow, and discharge. The land use distributions of the three watersheds are summarized in Table 2-7. The Tenkiller watershed is the most developed and contains the most pasture and hay. All three have a large fraction forested. No records indicated extensive silviculture or industrial activities in any of the watersheds.

[6] Although there is no one "perfect" comparison to Lake Tenkiller that has *all* of the same characteristics, but little or no poultry within the basin, attempts were made to find comparable watersheds that had important characteristics similar to that of the Lake Tenkiller watershed.

Table 2-7. Land use distribution for Lakes Tenkiller, Hugo, and Sardis Watersheds.

Land Cover	Tenkiller	Hugo	Sardis
Open Water	1.5%	2.6%	8.0%
Developed Open Space	5.6%	2.9%	1.5%
Developed, Low Intensity	2.1%	0.3%	0.1%
Developed, Medium Intensity	0.7%	0.1%	0.0%
Developed, High Intensity	0.3%	0.0%	0.0%
Barren Land	0.1%	0.1%	0.0%
Deciduous Forest	41.5%	33.7%	44.8%
Evergreen Forest	1.2%	23.3%	12.8%
Mixed Forest	0.5%	7.4%	9.0%
Shrub/Scrub	0.5%	1.6%	1.6%
Grassland/Herbaceous	3.4%	8.6%	5.6%
Pasture/Hay	42.0%	18.0%	15.4%
Cultivated Crops	0.1%	0.2%	0.0%
Woody Wetlands	0.6%	1.2%	1.1%
Emergent Herbaceous Wetlands	0.0%	0.1%	0.0%

Land cover information from 2001 Multi-Resolution Land Cover dataset.

An important characteristic pertinent to the comparison is the ratio of the size of the watershed to the size of the reservoir. This watershed to water surface area ratio is a measure of the area of land contributing runoff per unit area of reservoir. A higher value means more potential for water quality issues because relatively more land is contributing nutrients and bacteria to the lake. Given the importance of the watershed to water surface area ratio, comparisons to Lake Tenkiller need to be made in light of differences in these ratios.[7] The watershed to lake area ratios of Lakes Tenkiller, Hugo, and Sardis are 82.3, 81.4, and 12.3, respectively (Table 2-8). These ratios indicate that Tenkiller and Hugo undergo similar areal loading, while Sardis experiences significantly less.

[7] It should be noted that Drs. Cooke and Welch identify the watershed to water surface area ratio as an important differentiation between lakes and run-of-the-river reservoirs on page 9 (last paragraph) of their report. However, they ignore this characteristic when comparing Broken Bow to Lake Tenkiller. Broken Bow has a watershed to water surface area ratio of 37, while Lake Tenkiller's is 82. See Horne (2009) and Sullivan (2009) for further discussion concerning the inappropriateness of using Broken Bow as a comparative watershed to Lake Tenkiller.

Table 2-8. Comparison of various watershed characteristics among Lakes Tenkiller, Hugo, and Sardis.

Lake	Storage Conservation Control Pool (acre-ft)	Watershed Area (acre)	Water Surface Area (acre)	Watershed/ Water Surface Area Ratio	Average Depth (ft)
Lake Tenkiller	654,100	1,052,800	12,800	82.3	51.1
Lake Hugo	166,954	1,093,760	13,440	81.4	12.4
Lake Sardis	274,333	167,040	13,610	12.3	20.2

Hugo and Sardis have fewer poultry counts per unit area than Tenkiller (Table 2-9). The Tenkiller watershed contains approximately 213 animal units of poultry per square mile. The Hugo and Sardis watersheds contain seven and less than one animal units per square mile, respectively. The cattle populations in the Tenkiller, Hugo, and Sardis watersheds are 42, 28, and 25 animal units per square mile and the swine populations are 7, 2, and 6 animal units per square mile, respectively. The Tenkiller watershed contains the greatest density of people; 163 per square mile compared to 12 and 8 in Hugo and Sardis, respectively.

Table 2-9. Population counts for the Tenkiller, Hugo, and Sardis watersheds.

Lake	Active Poultry Houses per sq mi	2002 Cattle Animal Units per sq mi[1]	2002 Swine Animal Units per sq mi[1]	2005 Human Population per sq mi
Lake Tenkiller	1.2 (1.1)[2]	106	18	163
Lake Hugo	0.02	28	2	12
Lake Sardis	<0.01	25	6	8

Notes:

1. Poultry, cattle, and swine animal units acquired through personal communication with Raleigh Jobes.

2. Number of active poultry houses per Plaintiffs' consultant J. Berton Fisher. Number of active poultry houses per defendants' contract growers in parentheses.

A review of USEPA Water Discharge Permits revealed no significant point-source dischargers in either the Hugo or Sardis watersheds. Point-source dischargers are direct contributors to the nutrient loads in a watershed. The absence of significant discharges not only eliminates anomalous nutrient sources in the comparative watersheds, but further supports the assertion that there are comparable or less human populations and industry in the Hugo and Sardis watersheds as compared to the Illinois River Watershed.

Run-of-the-river reservoirs typically have lacustrine, transitional, and riverine zones. Generally, these different zones have differing water quality. Hugo and Sardis are somewhat unique in that the transition from river to lake occurs over a short distance and these lakes lack the type of riverine zone seen in Tenkiller (Figures 2-18a through 2-18c). When comparing the water quality among these three lakes, it is critically important that comparisons are made for like sections.

A reservoir's residence time can influence water quality. If the residence time is short (i.e., less than about two months; Baker and Dycus 2006), the loss of phytoplankton with the water flowing out of the reservoir can limit the maximum phytoplankton concentration in the reservoir. Table 2-10 shows an estimate of the residence times of the three reservoirs. Because Lake Hugo's residence time is low enough to potentially impact maximum phytoplankton concentrations (i.e., maximum chlorophyll-a concentrations), it needs to be considered when comparing Lake Hugo to the other two reservoirs.

Table 2-10. Estimate of residence times for Lakes Tenkiller, Hugo, and Sardis.

Reservoir	Age (yrs)	Volume[a] (ac-ft)	Average Depth (ft)	Approx. Long-Term Average Inflow (cfs)	Period of Record	Estimated Residence Time (months)[b]			
						Whole Lake	Riverine	Transitional	Lacustrine
Hugo	33	157,700	11.9	2,100	1995-2007	1.3	0.1	0.4	0.7
Sardis	27	274,330	20.2	325	1995-2007	14.2	-	2.5	11.7
Tenkiller	56	654,100	50.7	1,245	1997-2007	8.8	0.3	1.3	7.3

[a] At normal pool elevation.
[b] At long-term average inflows.
Long term average inflow for Hugo, Sardis, and Broken Bow determined from United States Army Corps of Engineers (USACE) charts, for Tenkiller, used average United State Geological Survey (USGS) flows for Baron Fork, Caney Creek, and Illinois River at Talequah.

Water quality was compared in two ways. The phytoplankton concentrations, measured as chlorophyll-a, total phosphorus, and SRP concentrations in the upstream sections provide some sense to the potential impact of point and non-point sources of phosphorus in the watershed. The chlorophyll-a, total phosphorus, and SRP concentrations, dissolved oxygen profiles in the lacustrine sections, and chlorophyll-a Trophic State Index (TSI) values provide evidence of the water quality impacts resulting from the watershed loads.

Figure 2-19 and Table 2-11 show the average surface concentrations of chlorophyll-a, total phosphorus, and SRP during the summer season (May through September) in the transitional section of each lake. The chlorophyll-a concentrations in the transitional sections of Lakes Hugo, Sardis, and Tenkiller during 2003 and 2005, where contemporaneous data are available, were similar in magnitude. These transitional section concentrations in 2003 and 2005 were 9.2, 7.0, and 8.0 μg/L and 11.0, 7.4, and 15.6 μg/L, respectively. Similar transitional section chlorophyll-a concentrations indicate that despite the smaller poultry populations in the Hugo and Sardis watersheds, a shorter residence time in Lake Hugo, and the lower lake to watershed ratio of Lake Sardis, the three lakes exhibit similar potential impact from their respective watersheds. This conclusion is further supported by similar total phosphorus and SRP concentrations in the transitional sections of the three lakes from 2003 and 2005. The average transitional section total phosphorus concentrations in Lakes Hugo, Sardis, and Tenkiller in 2003 and 2005 were 0.08, 0.02, and 0.17 mg/L and 0.08, 0.03, and 0.02 mg/L, respectively. The average transitional section SRP concentrations in 2003 and 2005 were 0.03, 0.01, and 0.09 mg/L and 0.03, 0.01, and <0.01, respectively.

Table 2-11. Summer surface average and chlorophyll-a, total phosphorus, and SRP concentrations in the transitional section of Lakes Hugo, Sardis, and Tenkiller from 2003 and 2005.

Parameter	Year	Lake	Number of Observations	Average	Minimum	Maximum	Units
Chlorophyll-a	2003	Hugo	5	9.2	4.9	13.0	mg/L
		Sardis	2	7.0	6.7	7.3	mg/L
		Tenkiller	13	8.0	2.9	33.2	mg/L
	2005	Hugo	3	11.0	8.0	13.0	mg/L
		Sardis	2	7.4	6.4	8.4	mg/L
		Tenkiller	25	16.2	8	32.3	mg/L
Total Phosphorus	2003	Hugo	6	0.077	0.068	0.091	mg/L
		Sardis	4	0.017	0.010	0.023	mg/L
		Tenkiller	5	0.171	0.025	0.310	mg/L
	2005	Hugo	6	0.081	0.072	0.093	mg/L
		Sardis	2	0.028	0.027	0.028	mg/L
		Tenkiller	15	0.023	0.003	0.033	mg/L
Soluble Reactive Phosphorus	2003	Hugo	6	0.031	0.016	0.043	mg/L
		Sardis	4	0.009	0.007	0.011	mg/L
		Tenkiller	6	0.090	0.010	0.190	mg/L
	2005	Hugo	6	0.033	0.024	0.043	mg/L
		Sardis	2	0.011	0.010	0.011	mg/L
		Tenkiller	15	0.002	0.001	0.005	mg/L

An analysis of the average surface concentrations of chlorophyll-a, total phosphorus, and SRP during the summer season (May through September) was also performed for the lacustrine section of the three lakes (Figure 2-20 and Table 2-12). Generally, the total phosphorus and SRP concentrations are lower in the lacustrine section of each lake as compared to upstream sections due to the settling of nutrients to the lake floor and phytoplankton utilization of the nutrients upstream of the lacustrine section. Chlorophyll-a concentrations are generally lower in the lacustrine section because nutrient concentrations are lower.

Table 2-12. Summer surface average total phosphorus, SRP, and chlorophyll-a concentrations in the lacustrine section of Lakes Hugo, Sardis, and Tenkiller from 2003 and 2005.

Parameter	Year	Lake	Number of Observations	Average	Minimum	Maximum	Units
Chlorophyll-a	2003	Hugo	4	5.5	2.6	10.6	µg/L
		Sardis	3	5.8	4.5	6.5	µg/L
		Tenkiller	26	4.8	1.2	9.9	µg/L
	2005	Hugo	2	9.0	6.0	12.0	µg/L
		Sardis	2	7.3	6.6	8.1	µg/L
		Tenkiller	47	11.1	4.0	36.8	µg/L
Total Phosphorus	2003	Hugo	4	0.060	0.040	0.081	mg/L
		Sardis	6	0.017	0.010	0.037	mg/L
		Tenkiller	13	0.146	0.011	0.420	mg/L
	2005	Hugo	4	0.068	0.051	0.090	mg/L
		Sardis	4	0.022	0.005	0.028	mg/L
		Tenkiller	24	0.013	0.008	0.027	mg/L
Soluble Reactive Phosphorus	2003	Hugo	4	0.027	0.016	0.038	mg/L
		Sardis	6	0.008	0.006	0.010	mg/L
		Tenkiller	17	0.073	0.005	0.170	mg/L
	2005	Hugo	4	0.031	0.019	0.046	mg/L
		Sardis	4	0.009	0.005	0.010	mg/L
		Tenkiller	22	0.003	0.001	0.013	mg/L

Identical comparisons and trends were apparent in the lacustrine sections as in the transitional sections of the three lakes. The 2003 and 2005 average summer surface chlorophyll-a and nutrient concentrations in the lacustrine sections of the three lakes were similar and lacustrine section nutrient concentrations in Lake Tenkiller decreased from 2003 to 2004. Lakes Hugo, Sardis, and Tenkiller average 2003 and 2005 summer surface chlorophyll-a lacustrine section concentrations were 5.5, 5.8, and 4.8 µg/L and 9.0, 7.3, and 11.1 µg/L, respectively. Lacustrine section total phosphorus concentrations were 0.06, 0.02, and 0.15 mg/L and 0.07, 0.02, and 0.01 mg/L, and SRP concentrations were 0.03, 0.01, and 0.07 mg/L and 0.03, 0.01, and <0.01 mg/L, respectively. These results further indicate similar water quality in the three lakes despite the lower poultry populations in the Hugo and Sardis watersheds and existing conditions in Lakes Hugo and Sardis that should improve water quality as compared to Lake Tenkiller (shorter residence time and lower watershed to lake ratio, respectively).

Figure 2-21 shows dissolved oxygen profiles in the lacustrine sections of Lakes Tenkiller, Hugo, and Sardis and the Plaintiff's comparison reservoir, Lake Broken Bow. The profiles were all taken during July and August. The four lakes have relatively high dissolved oxygen concentrations in the top 5 m and then trend toward zero dissolved oxygen near 10 m depth. Data are not available in Lake Sardis below 8 m and Lake Hugo has a relative shallow average depth, but the trend of the data appears similar for all four reservoirs. These dissolved oxygen profiles indicate that all of the reservoirs experience the common phenomena of dissolved oxygen depletion, even those whose watersheds have little poultry population. In fact, Sardis and Broken Bow, which have significantly lower watershed to water surface area ratios than the Tenkiller and Hugo, and thus potentially lower nutrient impacts, still show oxygen depletion in the bottom waters. In fact, most man-made run-of-river reservoirs will experience some level of dissolved oxygen depletion in the bottom waters, unless some other mechanism (such as wind mixing in shallow reservoirs) hinders dissolved oxygen depletion. In general, altering a natural system via dam construction inevitably results in water quality issues. Consequently, thermal stratification and resulting low dissolved oxygen levels in deeper waters is normal for run of the river reservoirs (Thornton et al. 1990)

Finally, chlorophyll-a TSI values were calculated for each section and the entire lake of Lakes Tenkiller, Hugo, and Sardis for the summer of 2005 (Figure 2-22). Trophic State Index provides a "rule-of-thumb" measure of the tropic status of the reservoir. The Oklahoma Water Resources Board (OWRB) uses chlorophyll-a TSI to assess what lakes in Oklahoma are eutrophic (or hypereutrophic) and potentially need to be managed to control algae. The TSI values calculated from a compilation of all available data are similar to the values found in Oklahoma's Beneficial Use Monitoring Program (BUMP) – Lakes Sampling, 2006-2007 Draft Report (OWRB 2007; eutrophic or borderline eutrophic). These results further support the existence of similar water quality issues in the three lakes, regardless of their poultry populations or conditions in Lakes Hugo and Sardis that should mitigate water quality impacts (shorter residence time and lower watershed to lake ratio, respectively).

Water quality issues in watersheds with low poultry populations relative to the Illinois River Watershed supports the conclusion that poultry litter is not the primary reason for water quality issues that exist in Lake Tenkiller. There are other factors affecting water quality in Lakes Tenkiller, Hugo, and Sardis. These include:

1. urban and rural development which increases impervious cover, lawn and golf course fertilization, wastewater treatment plant (WWTP) discharges, and the number of septic systems in the watershed (Nelson et al. 2002; Soerens 2003; Sonoda 2007);
2. deforestation and related erosion (Perry et al. 1999; Zheng 2005; Grip 2008; Grip 2009);
3. row crop synthetic fertilizers and related erosion (Sharpley and Smith 1990; Sharpley et al. 2003; Wortmann 2005);
4. other livestock operations such as cattle and swine (USDA 2003; Shaffer 2005; Wortmann 2005; Beede 2007); and
5. inputs from humans during recreational use (see Jarman 2008 for discussion).

Finally, and most importantly, altering a natural system via dam construction inevitably results in water quality issues. These water quality issues arise due to restricting sediment flux out of a watershed and decreasing the potential and kinetic energy of the system, which increases residence time in the water body and thus promotes growth of phytoplankton.[8]

2.9 WASTEWATER TREATMENT PLANTS APPEAR TO BE THE MOST IMPORTANT SOURCE OF BIOAVAILABLE PHOSPHORUS TO THE SYSTEM

Many wastewater treatment plants in the Arkansas and Oklahoma portions of the Illinois River Watershed installed significant upgrades within the past decade, the majority of which were in place by 2004 (Jarman 2008). Improvements have been seen in water quality

[8] Lakes Hugo and Sardis watersheds do not have significantly more urbanization, human population, or other animal populations compared to Lake Tenkiller. Consequently, the water quality issues observed in Lakes Hugo and Sardis even with the lower poultry populations can not be attributed to just urbanization, deforestation, or other animal populations.

immediately downstream of these facilities, and in some cases the water quality improvements have been noted far downstream in the wider Illinois River Watershed.

Wastewater treatment plants and their impact on Illinois River waters have been studied for numerous years. Haggard et al. (2003) and Ekka et al. (2003) indicate that base flow concentrations of phosphorus were elevated for streams receiving WWTP discharges. Haggard (2005) attributes decreased dissolved phosphorus concentrations in Spring Creek, and downstream in Osage Creek and the Illinois River, to upgrades to the Springdale municipal WWTP. Arkansas Department of Environmental Quality (ADEQ; 2008a) notes decreases in phosphorus concentrations in Siloam Springs, Sager Creek, and Little Sugar Creek over the past decade, in conjunction with treatment plant upgrades. Arkansas Water Resources Center (AWRC 2007) associated reduced total phosphorus base flow loads downstream of Siloam Springs to reduced wastewater treatment plant effluent loads, and found a strong correlation.

WWTP impacts continue to be seen in the water bodies in the Illinois River Watershed. Twenty-two percent of the impaired water bodies in the Oklahoma portion of the watershed include 'municipal point sources' as potential causes of the impairment (ODEQ 2008). 8.1 miles of Sager Creek remain impaired due to municipal point sources (ADEQ 2008b).

There are nine notable WWTPs that discharge to the streams of the Illinois River Watershed. Three are in Oklahoma and six are in Arkansas. Information about these plants is presented in Table 2-13.

Table 2-13. Wastewater treatment plants discharging to the Illinois River Watershed.

Plant	State	Receiving Water	Connection to the Illinois River	Average Total Phosphorus Load 2004 to 2007 (kg/yr)
Prairie Grove	AR	Unnamed tributary of Muddy Fork	Muddy Fork	2,000
Fayetteville – West	AR	Mud Creek (2004 – June 2007) Goose Creek (July 2007 – present)	Clear Creek Goose Creek	2,300
Springdale	AR	Spring Creek	Osage Creek	11,300
Rogers	AR	Osage Creek	Osage Creek	5,700
Siloam Springs	AR	Sager Creek	Flint Creek	13,000
Tahlequah	OK	Tahlequah Creek	Tahlequah Creek	1,200
Lincoln	AR	Unnamed tributary of Bush Creek	Baron Fork	270
Westville	OK	Shell Branch of Baron Fork	Baron Fork	330
Stillwell	OK	Caney Creek	Caney Creek	900

In total, over the period from 2004 to 2007 these plants discharged an average of almost 37,000 kg of phosphorus per year to the streams of the Illinois River Watershed, not counting any spikes in discharge that may have occurred due to plant upsets or short-circuiting during storm events (Jarman 2008). Much of the phosphorus entering the streams from these plants is dissolved and most of the dissolved phosphorus is reactive (i.e., SRP), the form that stimulates plant growth. This fact is evident in Figure 2-23, which shows the fraction dissolved and fraction of dissolved that is SRP for phosphorus measurements conducted by the Plaintiffs on WWTP effluent.

The influence of the WWTPs is evident in the spatial pattern of phosphorus concentrations in the rivers and streams of the Illinois River Watershed, as shown in Figure 2-24a. The highest total phosphorus concentrations (typically red or orange symbols) are found almost always just downstream of WWTPs (yellow diamonds in the figure). Moving further downstream there is typically a downward trend in concentrations indicated by the transition to green, light blue and finally dark blue symbols. High concentrations occur at a few stations remote from WWTPs, but the only organized spatial patterns are tied to the WWTPs.[9] A similar pattern exists for SRP, which is shown in Figure 2-24b.

[9] The location of the wastewater treatment facility in Watts, OK is also indicated on these figures. This is a retention and land application facility and is not permitted to discharge, however at least one release is documented (Jarman 2008). Sampling data from the Illinois River immediately downstream of the Watts facility is not available,

A more quantitative examination of the spatial patterns is presented in Figure 2-25, which shows the upstream to downstream trend in SRP concentrations in the Illinois River for three time periods (1998 to 2000; 2001 to 2003; 2004 to 2008). Red arrows indicate the locations where major tributaries enter the Illinois River. Moving from upstream to downstream, there is a gradual increase in SRP concentration from levels less than 0.01 mg/L to about 0.03 mg/L just above Muddy Fork (data only in the 2004-2008 period). The two sampling locations between Muddy Fork and Osage Creek exhibit similar concentrations in the range of 0.03 to 0.05 mg/L. The first station downstream of Osage Creek has concentrations in the neighborhood of 0.15 mg/L, a substantial increase from the nearest upstream station. This increase suggests that Osage Creek is an important source of SRP to the Illinois River. The reach between Osage Creek and Lake Frances shows increases in the two earlier time periods (though not statistically significant) and a statistically significant[10] decrease in the latest period. Concentrations generally decline between Lake Frances and Lake Tenkiller reaching about 0.07 to 0.09 mg/L just above Lake Tenkiller. The locations where these samples were collected are identified on Figure 2-26.

Given the apparent importance of Osage Creek, the spatial pattern in this creek and its tributaries was examined. Focusing on the 2004-2008 period (Figure 2-27), which has the best spatial coverage, and August 2006 (Figure 2-28) to provide a synoptic view, it is apparent that the influence of Osage Creek on SRP in the Illinois River is due to WWTPs. Beginning on Spring Creek, SRP concentrations are less than 0.1 mg/L upstream of the Springdale WWTP and about 0.45 mg/L just downstream of the plant. On average, levels decline to about 0.2 mg/L just upstream of the confluence with Osage Creek, although they are at 0.35 mg/L in August 2006. In Osage Creek, the concentration is about 0.01 mg/L upstream of the Rogers WWTP and 0.25 mg/L downstream of the plant. There is a drop to about 0.15 mg/L just upstream of the confluence with Spring Creek and an increase to close to 0.2 mg/L downstream of the confluence. Just above the confluence with Illinois River the concentration is about 0.12 mg/L (0.2 mg/L in August 2006). Similar patterns are shown in data measured before 2004

[10] Statistical significance inferred when differences fall outside the 2 standard error range indicated by the error bars around the mean values.

(Figure 2-29), indicating that historically, WWTP discharges had an influence on the phosphorus concentrations in the rivers and streams. See Figure 2-26 for sampling locations.

Wastewater treatment plants impact phosphorus concentrations in the Illinois River every day, whereas most other sources (except perhaps septic tanks) contribute only during runoff events that occur periodically and somewhat infrequently during the summer season when phosphorus impacts water quality. In fact, the amount of phosphorus in the Illinois River under base flow conditions corresponds to the amount that entered upstream from WWTPs, indicating that the WWTPs are the dominant source of phosphorus during base flow. This correspondence is shown in Figure 2-30, which displays the distribution of base flow phosphorus loadings measured by the United States Geological Survey (USGS) in the river at monitoring stations at Watts and Tahlequah and shows as vertical lines the average daily loading from the WWTPs. The average load from the WWTPs matches the central tendency base flow load in the river. The variability in the river around the central tendency likely reflects the day-to-day variability in WWTP load.

The 2004-2006 average daily wastewater treatment plant total phosphorus loads were also compared to 2004-2006 Illinois River and tributary average daily total phosphorus loads under base flow (WWTP data for 2007 were incomplete, therefore 2007 is not shown). Available daily flow and total phosphorus data from USGS gauging stations at Watts, Tahlequah, Baron Fork, and Caney Creek were used to estimate average daily total phosphorus loads with LOADEST, a program that estimates average loads through a rating curve method (Runkel et al. 2004).[11] As shown in Figure 2-31, the wastewater treatment plant loads (per Jarman 2008) are reasonable matches to the base flow loads in 2005 and 2006. The treatment plant loads appear lower than the in-river base flow loads in 2004 when frequent and significant high flow events potentially biased the estimation of base flow (i.e., some high flows identified as base flows may have included surface runoff) and the elevated base flows may have introduced a greater load from septic systems (see Figure C-1 to note the high base flows in

[11] LOADEST estimated daily loads with available paired daily average flow and total phosphorus data. Daily average flow data were used because instantaneous flow data were not available at all locations. Daily average total phosphorus loads are averages of daily total phosphorus loads estimated by LOADEST. LOADEST load estimates were generated using the model's Method 8 and separate rating curves were produced for each year.

2004). Note, the locations labeled as Baron Fork and Caney Creek in Figure 2-31 refer to the points in the Illinois River where the Baron Fork and Caney Creek tributaries meet the Illinois River.

In contrast to base flow phosphorus, runoff–associated phosphorus is not present in the river on a day-to-day basis. In addition, much of the runoff phosphorus load is associated with particulate matter, which would have little direct impact on water quality (it can exert an influence via recycle from sediments). This fact is illustrated in Figure 2-32, which shows the fraction of total phosphorus that is particulate in relation to river flow (particulate phosphorus is calculated by subtracting dissolved phosphorus from total phosphorus). A consistent increase with increasing flow is evident.

The particulate phosphorus associated with runoff events will only settle out of the water column when the river velocity is less than about 15 miles/day (Ziegler et al. 2000). Due to the high velocities characteristic of the Illinois River within Oklahoma[12] (Figure 2-33), little of the particulate phosphorus settles in the river. Much of the runoff particulate phosphorus likely settles out in Lake Tenkiller. This sediment phosphorus might later contribute to phosphorus levels in the lake if it fluxes out of the sediment, but in general it has limited bioavailability (see section 2-10).

During the summer season (May to September), the river experiences runoff conditions only about 20% of the time.[13] Due to the short duration of runoff events, their relative infrequency, and the nature of the phosphorus, run-off associated phosphorus has little impact on water quality, except possibly within Lake Tenkiller.

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[12] River velocities determined using Manning's Equation with a Manning's roughness coefficient of 0.04; slope determined from USGS gage heights (when available) or Google map topographic elevations, and river distances determined from GIS using Environmental System Research Institute data; depths of water surface determined from USGS depth data coincident with average summer-time flow rates at each USGS gage location. Riverine portion of lake velocities determined by dividing summer-time average flow rate just downstream of Baron Fork by the approximated cross section of the riverine portion of the lake between Baron Fork and LK04; distance from Baron Fork to LK04 determined from GIS.

[13] The contributions of base flow and runoff flow to the river hydrograph was determined using a base flow separation methodology described in Appendix C.

2.10 LAKE SEDIMENT PHOSPHORUS IS A MINOR SOURCE OF BIOAVAILABLE PHOSPHORUS

Phosphorus enters Lake Tenkiller with point source (i.e., WWTP) dominated base flow, non-point source dominated high flow, and groundwater discharge. The eventual fate of phosphorus once it enters the lake varies depending on the specific forms of phosphorus present. Any dissolved phosphorus will either remain in the water column of the lake or be discharged downstream via the dam. SRP can be taken up by algae and eventually converted to particulate phosphorus. Particulate forms of phosphorus may remain in the water column, be discharged downstream via the dam, or settle to the bottom and become incorporated in the sediment.

Phosphorus movement into, through, and out of the reservoir is illustrated as a conceptual diagram (Figure 2-34). Of note is the summertime stratification of the reservoir into 2 layers. Reservoirs such as Lake Tenkiller tend to be completely mixed, with similar temperature and chemical constituents throughout, during the winter (Lewis 1983). The warming of the water surface during early spring initiates separation of the lake water into distinct layers. As the temperature of the surface water exceeds 39°F, its density declines.[14] This more buoyant water remains near the top of the water column. Warm, bouyant water toward the lake surface becomes the summer epilimnion, or top layer of the water column. Dissolved oxygen levels increase due to exposure to the atmosphere (USEPA 2000). Light is available for photosynthesis, and the potential for further increases in dissolved oxygen. The presence of algae will be indicated by increased levels of chlorophyll-a.

The colder, denser layer that forms at the bottom of the lake is called the hypolimnion, where low temperature and lack of light penetration, inhibit algae growth. Due to differences in density, these two layers do not mix, and there is little transport of dissolved chemical constituents (including oxygen) between the epilimnion and the hypolimnion.

With colder weather in the autumn, the epilimnion water cools, and increases in density. When the density of the epilimnion water exceeds the density of the hypolimnion, a fall turnover

[14] Water reaches maximum density at 39°F. Above or below this temperature, water will be more buoyant.

occurs; epilimnion waters tend to sink and hypolimnion waters tend to rise. The temperature and density of the water is closer to uniform, so individual layers do not remain after the turnover. The lake remains completely mixed during the winter, and the cycle repeats in the spring.

Since 2004, about 205,000 kg/yr total phosphorus entered Lake Tenkiller from the Illinois River, Baron Fork, and Caney Creek (Bierman 2009). About 30,000 kg/year exited via Tenkiller Dam.[15] The remaining 175,000 kg/year of phosphorus was incorporated into the bottom sediments, with minimal if any impact on algae levels in the lake, as described below.

During the summer months, dissolved oxygen is depleted in the hypolimnion. When dissolved oxygen is very low or zero in the hypolimnion, some of the phosphorus in the sediments can return to the water column as dissolved phosphorus, largely in the form of SRP. This flux increases the summer SRP concentration in the hypolimnion, and contributes phosphorus to the surface waters when the lake overturns in late fall. In Lake Tenkiller, the hypolimnetic SRP mass increases by approximately 3,000 kg[16] during the summer, but this mass is not large enough to have a material impact on the epilimnion SRP concentrations when the lake turns over (top row of graphs in Figure 2-35); SRP concentrations never get higher than 0.01 mg/L in the epilimnion.

It should be noted that Figure 2-35 shows that ***total*** phosphorus increases in the hypolimnion during the summer. In fact, this increase is quite significant compared to the increase in SRP and other forms of dissolved phosphorus (not shown), indicating that the increase in phosphorus is comprised mostly of particulate-bound phosphorus. This is likely the result of river water plunging in the vicinity of LK-04 and "pulling" chlorophyll-a and suspended sediment from the surface waters into the bottom waters (see Section 4.2 for further discussion of the plunging river water in the vicinity of LK-04). It is not likely caused by resuspension because one would expect a more random or event-based (i.e., scour events) signature.

[15] Estimated using a long-term mean of hypolimnion phosphorus concentrations and the 1994 – 2007 United States Army Corps of Engineers record of water release at the dam.
[16] Mass of SRP in the hypolimnion was found using the Plaintiffs' data collected in the deep waters of Lake Tenkiller, combined with an estimate of the hypolimnion volume from Dr. Wells' lake model bathymetry.

Although non-point-source phosphorus loads contribute to sediment phosphorus in Lake Tenkiller, much of this phosphorus is locked in the sediment and does not contribute to algae growth.

SECTION 3
PHOSPHORUS HAS MINIMAL IMPACT ON THE ILLINOIS RIVER IN OKLAHOMA

3.1 SUMMARY OF DETAILED FINDINGS

- Phosphorus is not causing excessive growth of phytoplankton in the Illinois River.
- Benthic algae are rarely at densities considered a nuisance.
- The frequency of dissolved oxygen criteria violation in the Illinois River are minimal and can not be connected to any one land use.
- The fisheries in the Illinois River in Oklahoma are not damaged.

3.2 PHOSPHORUS IS NOT CAUSING EXCESSIVE GROWTH OF PHYTOPLANKTON IN THE ILLINOIS RIVER

Like other photosynthesizing life-forms, phytoplankton growth depends on light, temperature, and nutrients. The Illinois River in Oklahoma contains enough nutrients for phytoplankton to grow. Bioavailable phosphorus (BP; measured as soluble reactive phosphorus), which is in shorter supply than bioavailable nitrogen (BN; measured as ammonia plus nitrate), is typically found at concentrations close to 100 µg/L; about five times above levels at which growth begins to slow appreciably. Yet, phytoplankton concentrations in the river are relatively low, typically peaking at levels much less than 10 µg chlorophyll-a/L. The fact is illustrated in Figure 3-1, which shows chlorophyll-a and soluble reactive phosphorus at Watts, OK and Tahlequah, OK stations that are representative of the upper and lower portions of the river in Oklahoma. Phosphorus is not causing excessive growth of phytoplankton in the river.

Phytoplankton concentrations in the river are low despite the availability of phosphorus because water flows too quickly through the river for phytoplankton to grow. The river has a relatively steep slope, dropping about 230 ft. between Watts and Tahlequah (based on the USGS datum at the Watts and Tahlequah flow gages). Under a typical summer flow of 400 cfs at Tahlequah, the river is about three feet deep, it moves at about 2 ft. per second and it takes about

1.5 days to travel from Watts to Tahlequah (See Figure 2-31).[17] If conditions are perfect for growth, phytoplankton might increase by a factor three in 1.5 days.[18] This means that if chlorophyll-a starts out at 2 μg/L at Watts, under ideal conditions it might increase to about 6 μg/L by Tahlequah, not accounting for dilution that would occur as water enters the river between Watts and Tahlequah. There is insufficient time to reach levels that affect the aesthetic quality of the water, which are certainly greater than 10 μg/L. A study of 116 Florida lakes (Hoyer et al. 2004) found that the chlorophyll-a level of water whose algal content was perceived to slightly impair swimming and aesthetic enjoyment averaged 14 μg/L and the chlorophyll-a level of water whose algal content substantially reduced the desire to swim averaged 17 μg/L. A similar study of Texas lakes found that the chlorophyll-a level associated with a substantial reduction in the desire to swim averaged 27 μg/L (Texas Water Conservation Association [TWCA] 2005). Consistent with these findings, the State of Minnesota uses chlorophyll-a levels of 20 μg/L for lakes and reservoirs in the Northern Lakes and Forests and North Central Hardwood Ecosystems and 30 μg/L for water bodies in the Western Corn Belt Plains and Northern Glacial Plains Ecosystems as thresholds for a nuisance algae bloom (MPCA 2004). The State of Oregon defines a nuisance algae bloom in a reservoir as a chlorophyll-a concentration of 15 μg/L, which is specified as the concentration representative of the average over a depth range from the surface to twice the Secchi depth and the time average for a three-month period (Oregon Department of Environmental Quality 2008). The Montana Department of Environmental Quality recommended a maximum chlorophyll-a concentration of 20 μg/L for wadeable streams in Montana's Hi-line region (Suplee 2004). In a study of over 200 North American and New Zealand streams and rivers, Dodds et al. (1998) suggested the mesotrophic-eutrophic boundary is at 20 μg/L.

[17] Velocity calculated using Manning's equation with a Manning's roughness coefficient of 0.04.

[18] Growth rate of 1.2/day estimated using the parameters used by Dr. Wells to describe phytoplankton growth, no nutrient limitation and a secchi depth of 4 meters. It is important to note that algae will move with water currents and therefore, rivers with high velocities will not experience maximum algae growth because the algae will not be "exposed" to the nutrients long enough in one place to reach maximum growth.

3.3 BENTHIC ALGAE ARE RARELY AT DENSITIES CONSIDERED A NUISANCE

Dr. Jan Stevenson, a Plaintiffs' consultant, cites two studies in his report indicating that benthic algae become a nuisance at densities greater than 10-15 μg chlorophyll-a/cm^2. Above these threshold densities filamentous species tend to dominate and cover greater than 20% of the stream bottom (Welsh et al. 1988). The USEPA reports that below 15 μg/cm^2 the aesthetic quality use will probably not be appreciably degraded by filamentous mats or other adverse effects attributed to dense mats of filamentous algae (USEPA 2000). Biggs (2000) recommended setting maximum algal biomass of 20 μg/cm^2 with a 30% maximum coverage of visible stream bed by filamentous algae for the protection of aesthetic and trout fishing values for rivers and streams in New Zealand. In a study of over 200 North American and New Zealand streams and rivers, Dodds et al. (1998) suggested the mesotrophic-eutrophic boundary of 20 μg/cm^2. In 2004, Montana Department of Environmental Quality recommended the several numeric criteria for wadeable streams in Montana's Hi-line region, a region covered mainly with semi-arid grasslands used extensively for livestock grazing and growing cereal grain crops. The criteria included maximum streambed cover by filamentous algae of 30% and benthic algae maximum density of 11 μg/cm^2 (Suplee 2004).

The measurements of benthic algae conducted in the Oklahoma portion of the Illinois River and its tributaries by the Plaintiffs' consultants, which are summarized as frequency distributions in Figure 3-2, show that nuisance densities are rare.[19] In summer 2006, the maximum density was 13.8 μg chlorophyll-a/cm^2 and about 95% of the stations had densities less than 10 μg chlorophyll-a/cm^2. In spring 2007, the maximum density was 33.5 μg chlorophyll-a/cm^2, but almost 90% of the stations had densities less than 10 μg chlorophyll-a/cm^2. Densities above 10 μg chlorophyll-a/cm^2 occurred principally in tributaries and frequently downstream of WWTPs. Only one station in the Illinois River in each sampling year had a value greater than 10. Higher values were prevalent in Spring Creek and Sager Creek

[19] The rarity of nuisance benthic algal blooms also invalidates Dr. Stevenson's use of 0.027 mg/L total phosphorus as a benchmark to understand when a particular river or stream in the Illinois River Watershed would have aesthetic issues or "damages". Nuisance levels of benthic algae are rarely measured, yet surface water concentrations of total phosphorus in the Illinois River are routinely above 0.027 mg/L. This fact promotes the establishment of a site-specific benchmark using the available data, as suggested in Stevenson et al. 2006 and Dodds et al. 1997.

as shown in Figure 3-3. On both tributaries, the higher values were found downstream of WWTPs; Siloam Springs on Sager Creek and Springdale on Spring Creek.

The influence of WWTPs on benthic algae is also evident in a USEPA Region 6 2003 study of diel dissolved oxygen variations upstream and downstream of WWTPs (Parsons and UA 2004). Diel dissolved oxygen variations downstream of the Prairie Grove WWTP on Muddy Fork are much greater than exist upstream (Figure 3-4a), indicating a high density of benthic algae. In contrast, little upstream to downstream change is evident around the Rogers WWTP on Puppy Creek (Figure 3-4b). A notable difference between the sites is the slope of the receiving stream; Puppy Creek slopes about 2 feet/mile, whereas Muddy Fork slopes about 1 ft/mile. The steeper slope of Puppy Creek probably means higher velocities, which could limit the density of benthic algae.

Dr. Stevenson examined percent cover by filamentous green algae in addition to benthic algae density. I was not able to replicate his presentation of these data (Figure 2.21 in his May 2008 report), but relying on his presentation, it appears that most stations had less than 30 percent cover. Reading from his graph, I estimate that 30 percent was exceeded at only 4 of 69 stations in 2006 and 27 of 70 stations in 2007. Not being able to replicate his presentation, I was unsure of the validity of the dataset in my possession and did not attempt to locate the high percent cover stations, but the density data suggest they would likely be in small tributaries downstream of WWTPs.

3.4 THE FREQUENCY OF DISSOLVED OXYGEN CRITERIA VIOLATIONS IN THE ILLINOIS RIVER ARE MINIMAL AND CAN NOT BE CONNECTED TO ANY ONE LAND USE

Drs. Cooke and Welch argue that low levels of dissolved oxygen have a strong negative impact on ecosystems of the water bodies of the Illinois River Watershed, and that much of the reduction in dissolved oxygen levels can be traced to land application of poultry litter. Dissolved oxygen data collected throughout the watershed refute this assertion.

Oklahoma regulations consider a stream to support the designated beneficial use of a cool water aquatic community if "no more than 10% of the samples from a stream are less than the screening level for DO" (OWRB 2008). As Figure 3-5 illustrates, for 2004-2007, the standard was met; only 3.4% of summer dissolved oxygen measurements, and 3.8% of dissolved oxygen measurements taken during the remainder of the year were below the associated criteria.

Of the 171 river and stream miles of the Illinois River Watershed that Oklahoma lists as not meeting water quality standards, only a 1.6 mile stretch of Flint Creek is listed as impaired due to dissolved oxygen (OWRB 2008). Nine potential sources are listed for the dissolved oxygen impairment of this stream segment.

Illinois River Watershed stream locations with sufficient dissolved oxygen data to assess water quality during 2004-2007 are indicated on Figure 3-6.[20] In 2007, 11% of the dissolved oxygen readings at the Flint Creek location were below the criteria. All other locations assessed had fewer than ten percent of the dissolved oxygen readings below the criteria for each year of the assessment. Land uses are also indicated on this map, and as can be seen, the majority of the land draining to locations with reduced dissolved oxygen is classified as deciduous forest or developed open space.

These data showing minimal dissolved oxygen violations, and the multiple potential sources of the dissolved oxygen impairments listed by the Oklahoma DEQ do not support the conclusion that poultry litter has impacted oxygen levels in the Illinois River Watershed.

3.5 THE FISHERIES IN THE ILLINOIS RIVER IN OKLAHOMA ARE NOT DAMAGED

In his report, Dr. Jan Stevenson evaluated fisheries in the Illinois River Watershed, from 37 locations in Arkansas and Oklahoma. His stated objective was "to document the injuries of fish species composition that are related to poultry house activities and nutrient pollution"

(Stevenson 2008; Section 4.1, p. 37). However, the analysis presented in his report fails to assess if the fisheries are actually injured, let alone injured due to poultry litter application and/or nutrient pollution.

Pollutants and other environmental stresses may simplify ecosystems by reducing the number of species present and by shifting the relative abundances of the surviving populations toward dominance by stress resistant species (Odum 1969; Woodwell 1970). The data collected in 2007 was intended to provide a basis to assess overall fish composition and abundance.[21] Most study sites contained species expected to occur within streams in the Ozark Highlands Ecoregion (with percids, cyprinids, and centrarchids typically most abundant [Dauwalter et al. 2003; Table 3-1]). The most common species collected in 2007 from the 37 Plaintiffs' locations were fluvial specialists such as stonerollers (*Campostoma spp*), cardinal shiner (*Luxilus cardinalis*), orangethroat darter (*Etheostoma spectabile*), and banded sculpin (*Cottus carolinae*). These four stream dwelling species prefer clear gravel bottom streams and require flowing water during some portion of their life history. Additionally, cardinal shiner is reported as one of the most intolerant fishes in Oklahoma of degradation to both water quality and habitat (Jester et al. 1992). Therefore, the presence of the cardinal shiner would indicate that water quality is not degraded. This species accounted for more than 2% of the overall abundance in 27 out of 37 locations (73%), and averaged 14% of the abundance at all locations (Table 3-1).

The overall composition and representativeness of species at each location provide additional insights regarding fishery health. We calculated Shannon-Weiner diversity and evenness for each location. Diversity values ranged from 1.01 to 2.58 with the two reference sites (Little Lee Creek RS-10003 and RS-10004)[22] at 2.09 and 2.11, respectively (Table 3-1). The lower diversity values, which may suggest some impact or may be due to smaller order streams being less diverse, are scattered throughout the watershed with no evident spatial patterns (Figure 3-7). Evenness was calculated to assess the relative spread of species and

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[20] Only locations with at least eight records in at least two years were considered. In addition, to ensure year-round oxygen status, only locations with at least one DO records in at least 3 quarters (three-month periods) were considered.

[21] Note: not all data used in this analysis were provided from the Plaintiffs' laboratory sheets. Additional data were used from Stevenson's considered materials; specifically: "Fish analysis.mdb" and "Database CDM 20080518.mdb."

evaluate if sites were dominated by one species. Values can range from zero for sites with one species dominant to one for sites were all species are found in equal numbers. Within the Illinois River Watershed, evenness ranged from 0.369 to 0.917; with values at the two reference sites of 0.753 and 0.656 (Table 3-1). While a few sites were dominated by one or two species, the majority of sites had fairly good representation of several stream species.

The index of biotic integrity (IBI) is a valuable metric that was developed to provide a straightforward and relatively quick method to assess local stream conditions based on the fish community (Karr et al. 1986). Fish integrate many trophic levels, providing a broad view of the biological community. The IBI is calculated and general descriptions given to each range of scores (e.g., good, fair, poor; see Chadwick 2009 for complete description of the IBI).

While initially developed for Midwestern streams, the IBI has been modified for several ecoregions throughout the United States, Mexico, and Europe. Recently, Dauwalter et al. (2003) developed an IBI for the Ozark Highlands Ecoregion in Arkansas. After review of the model, it was applied to the 37 locations in the Illinois River Watershed sampled in 2007. The final IBI is based on seven metrics representing taxonomic, trophic, reproductive, and health characteristics of fish asssemblages (Dauwalter et al. 2003).[23] Most of the final metrics were most significantly correlated with nutrients, chloride, land use, road densities, and sedimentation (Dauwalter et al. 2003), and should provide a robust method for assessing overall integrity.

Results of the IBI analysis within the Illinois River Watershed indicate most sites are in good condition (Table 3-1 and Figure 3-8). The majority of the sites rated as "good" are found in Oklahoma. To further evaluate the IBI score, comparisons were made between the IBI and local watershed characteristics, including:

- subwatershed area (Figure 3-9);
- poultry house density (Figure 3-10);
- road density (Figure 3-11);

[22] Note: the two reference sites are located outside of the Illinois River watershed.

- percent developed area (Figure 3-12);
- percent forested area (Figure 3-13);
- percent pasture area (Figure 3-14);
- density of WWTP discharges (Figure 3-15);
- distance to nearest road (Figure 3-16);
- distance to nearest urban land use classification (Figure 3-17); and
- distance to nearest poultry house (Figure 3-18).

There was no statistically significant relationship between the IBI value and any of these variables. For stations that had values below the minimum value for good scores (less than 60), points were scattered along the x-axis, rather than being clumped around any one value.

In summary, the fish community within the Illinois River Watershed is not highly degraded due to water quality impacts. While diversity is low in some locations, this is not unexpected due to the size of the streams (smaller streams will support fewer species). Stevenson also observed a direct relationship between fish species number and watershed size with fewer species in smaller watersheds (Stevenson 2008, Section 4.3.2.1., p. 40). There are limited data available on habitat parameters, so habitat quality can not be assessed at this time. However, it is possible that sites with lower IBI and/or diversity index scores may be more impacted by habitat availability than water quality degradation. Jester et al. (1992) reported that the majority of Oklahoma fish species are more sensitive to habitat degradation than they are to water quality degradation. Finally, the protocol used to sample fish may underestimate the diversity of fish within the watershed. Electrofishing consisted of sampling a habitat unit (e.g., riffle, pool) for three minutes (five minutes for boat shocking) and collecting stunned fish. In some cases, it appears that a second or third one- to three-minute period was sampled, although the exact protocols for this were not defined in the Standard Operating Procedures (SOP). It is fairly remarkable that the diversity within the watershed is as high as it is based on the low effort expended sampling each location. Diversity likely would be higher if more effort was expended

[23] Note: metric number 2 – percent with black spot or anomaly - was excluded due to insufficient data in the database.

at each site, especially in terms of the larger fish that more easily escape capture in a short period of time.

Table 3-1. Summary of species composition in the Illinois River Watershed based on the Plaintiff's 2007 data.

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
Ballard Creek	AR	RS-399	Campostoma spp.	298	73.0	3	1.03	0.447	57	Fair
		RS-399	Etheostoma spectabile	42	10.3					
		RS-399	Luxilus cardinalis	25	6.1					
		RS-399	Lepomis cyanellus	17	4.2					
		RS-399	Phoxinus erythrogaster	14	3.4					
		RS-399	Other (5 spp)	12	2.9					
	OK	BS-62A	Campostoma spp.	192	28.6	3	1.95	0.704	78	Good
		BS-62A	Cottus carolinae	127	18.9					
		BS-62A	Luxilus cardinalis	127	18.9					
		BS-62A	Etheostoma spectabile	84	12.5					
		BS-62A	Lepomis megalotis	62	9.2					
		BS-62A	Other (9 spp)	33	4.9					
		BS-62A	Noturus exilis	24	3.6					
		BS-62A	Lepomis macrochirus	23	3.4					
Flint Creek	AR	RS-160	Cottus carolinae	148	46.5	4	1.64	0.747	63	Good
		RS-160	Phoxinus erythrogaster	59	18.6					
		RS-160	Semotilus atromaculatus	36	11.3					
		RS-160	Etheostoma flabellare	19	6.0					
		RS-160	Campostoma spp.	17	5.3					
		RS-160	Etheostoma spectabile	16	5.0					
		RS-160	Catostomus commersoni	15	4.7					
		RS-160	Other (2 spp)	8	2.5					
	OK	RS-902	Cottus carolinae	117	35.2	4	1.74	0.641	74	Good
		RS-902	Campostoma spp.	102	30.7					
		RS-902	Luxilus cardinalis	45	13.6					
		RS-902	Other (9 spp)	23	6.9					
		RS-902	Noturus exilis	20	6.0					
		RS-902	Etheostoma spectabile	18	5.4					
		RS-902	Micropterus dolomieu	7	2.1					
		RS-421	Etheostoma spectabile	221	45.6	4	1.62	0.614	75	Good
		RS-421	Campostoma spp.	99	20.4					
		RS-421	Luxilus cardinalis	60	12.4					
		RS-421	Noturus exilis	45	9.3					
		RS-421	Semotilus	23	4.7					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
			atromaculatus							
		RS-421	Etheostoma punctulatum	20	4.1					
		RS-421	Other (8 spp)	17	3.5					
Upper Illinois River	AR	RS-234	Campostoma spp.	320	39.0	3	1.96	0.678	68	Good
		RS-234	Luxilus cardinalis	142	17.3					
		RS-234	Lepomis megalotis	94	11.5					
		RS-234	Other (11 spp)	75	9.1					
		RS-234	Lepomis cyanellus	70	8.5					
		RS-234	Etheostoma spectabile	65	7.9					
		RS-234	Pimephales notatus	36	4.4					
		RS-234	Etheostoma blennioides	18	2.2					
Middle Illinois River	OK	RS-757	Luxilus cardinalis	229	32.0	6	2.16	0.635	58	Fair
		RS-757	Lepomis megalotis	192	26.9					
		RS-757	Other (21 spp)	76	10.6					
		RS-757	Moxostoma erythrurum	68	9.5					
		RS-757	Lepomis macrochirus	37	5.2					
		RS-757	Pimephales notatus	33	4.6					
		RS-757	Dorosoma cepedianum	26	3.6					
		RS-757	Campostoma spp.	22	3.1					
		RS-757	Lepomis cyanellus	17	2.4					
		RS-757	Micropterus punctulatus	15	2.1					
Lower Illinois River	OK	RS-433A	Luxilus cardinalis	401	64.3	6	1.52	0.471	70	Good
		RS-433A	Notropis boops	66	10.6					
		RS-433A	Other (19 spp)	64	10.3					
		RS-433A	Lepomis megalotis	33	5.3					
		RS-433A	Campostoma spp.	26	4.2					
		RS-433A	Micropterus dolomieu	19	3.0					
		RS-433A	Pimephales notatus	15	2.4					
		RS-654	Pimephales notatus	153	18.1	6	2.58	0.767	62	Good
		RS-654	Notropis boops	127	15.1					
		RS-654	Luxilus cardinalis	94	11.2					
		RS-654	Lepomis megalotis	92	10.9					
		RS-654	Other (18 spp)	82	9.7					
		RS-654	Dorosoma cepedianum	64	7.6					
		RS-654	Dorosoma petenense	61	7.2					
		RS-654	Hypentelium nigricans	52	6.2					
		RS-654	Notropis nubilus	33	3.9					
		RS-654	Campostoma spp.	32	3.8					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
		RS-654	Lepomis macrochirus	28	3.3					
		RS-654	Moxostoma erythrurum	25	3.0					
Trib to Lower Illinois River	OK	RS-604	Luxilus cardinalis	587	51.2	4	1.2	0.502	69	Good
		RS-604	Etheostoma spectabile	305	26.6					
		RS-604	Campostoma spp.	209	18.2					
		RS-604	Other (8 spp)	46	4.0					
Unnamed tributary to Illinois River	OK	RS-772	Cottus carolinae	187	53.4	3	1.34	0.642	53	Fair
		RS-772	Phoxinus erythrogaster	81	23.1					
		RS-772	Campostoma spp.	30	8.6					
		RS-772	Semotilus atromaculatus	30	8.6					
		RS-772	Etheostoma spectabile	13	3.7					
		RS-772	Other (3 spp)	9	2.6					
Bush Creek	AR	BS-HF22	Phoxinus erythrogaster	69	24.3	3	2.11	0.8	68	Good
		BS-HF22	Campostoma spp.	51	18.0					
		BS-HF22	Cottus carolinae	49	17.3					
		BS-HF22	Etheostoma spectabile	42	14.8					
		BS-HF22	Semotilus atromaculatus	21	7.4					
		BS-HF22	Noturus exilis	13	4.6					
		BS-HF22	Other (4 spp)	11	3.9					
		BS-HF22	Etheostoma punctulatum	9	3.2					
		BS-HF22	Etheostoma flabellare	7	2.5					
		BS-HF22	Lepomis cyanellus	6	2.1					
		BS-HF22	Luxilus cardinalis	6	2.1					
Cincinnati Creek	AR	RS-392	Phoxinus erythrogaster	124	38.5	3	1.7	0.661	68	Good
		RS-392	Etheostoma spectabile	70	21.7					
		RS-392	Campostoma spp.	54	16.8					
		RS-392	Cottus carolinae	30	9.3					
		RS-392	Luxilus cardinalis	25	7.8					
		RS-392	Other (8 spp)	19	5.9					
		RS-386	Etheostoma spectabile	130	29.3		1.8	0.752	77	Good
		RS-386	Campostoma spp.	99	22.3	3				

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
		RS-386	Luxilus cardinalis	79	17.8					
		RS-386	Etheostoma punctulatum	61	13.8					
		RS-386	Cottus carolinae	39	8.8					
		RS-386	Semotilus atromaculatus	11	2.5					
		RS-386	Noturus exilis	10	2.3					
		RS-386	Phoxinus erythrogaster	10	2.3					
		RS-386	Other (3 spp)	4	0.9					
		BS-68	Campostoma spp.	169	29.9	4	1.69	0.641	75	Good
		BS-68	Luxilus cardinalis	168	29.7					
		BS-68	Etheostoma spectabile	117	20.7					
		BS-68	Noturus exilis	54	9.6					
		BS-68	Other (8 spp)	23	4.1					
		BS-68	Etheostoma punctulatum	17	3.0					
		BS-68	Semotilus atromaculatus	17	3.0					
Fly Creek	AR	BS-35	Campostoma spp.	344	45.1	3	1.41	0.551	74	Good
		BS-35	Etheostoma spectabile	257	33.7					
		BS-35	Luxilus cardinalis	76	10.0					
		BS-35	Other (8 spp)	40	5.2					
		BS-35	Cottus carolinae	26	3.4					
		BS-35	Lepomis cyanellus	20	2.6					
Muddy Fork	AR	RS-233	Lepomis megalotis	75	24.0	4	2.48	0.827	65	Good
		RS-233	Etheostoma spectabile	35	11.2					
		RS-233	Campostoma spp.	33	10.5					
		RS-233	Luxilus cardinalis	29	9.3					
		RS-233	Lepomis cyanellus	26	8.3					
		RS-233	Pimephales notatus	20	6.4					
		RS-233	Other (8 spp)	19	6.1					
		RS-233	Etheostoma blennioides	19	6.1					
		RS-233	Cottus carolinae	18	5.8					
		RS-233	Lepomis macrochirus	11	3.5					
		RS-233	Noturus exilis	10	3.2					
		RS-233	Etheostoma zonale	9	2.9					
		RS-233	Lepomis gulosus	9	2.9					
Spring Creek	AR	RS-121	Etheostoma spectabile	63	27.2	4	1.95	0.76	54	Fair
		RS-121	Campostoma spp.	45	19.4					
		RS-121	Luxilus cardinalis	43	18.5					
		RS-121	Noturus exilis	34	14.7					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
		RS-121	Lepomis megalotis	15	6.5					
		RS-121	Lepomis cyanellus	12	5.2					
		RS-121	Other (6 spp)	11	4.7					
		RS-121	Lepomis macrochirus	9	3.9					
Baron Fork	OK	RS-682	Cottus carolinae	181	51.4	4	1.53	0.597	66	Good
		RS-682	Campostoma spp.	77	21.9					
		RS-682	Etheostoma spectabile	27	7.7					
		RS-682	Luxilus cardinalis	23	6.5					
		RS-682	Other (7 spp)	17	4.8					
		RS-682	Noturus exilis	16	4.5					
		RS-682	Etheostoma punctulatum	11	3.1					
		RS-649	Campostoma spp.	410	37.2	6	1.83	0.574	78	Good
		RS-649	Luxilus cardinalis	334	30.3					
		RS-649	Other (18 spp)	100	9.1					
		RS-649	Etheostoma spectabile	92	8.4					
		RS-649	Cottus carolinae	83	7.5					
		RS-649	Noturus exilis	57	5.2					
		RS-649	Lepomis megalotis	25	2.3					
Bidding Springs	OK	RS-706	Luxilus cardinalis	68	23.7	2	2.1	0.714	77	Good
		RS-706	Etheostoma flabellare	64	22.3					
		RS-706	Campostoma spp.	50	17.4					
		RS-706	Etheostoma spectabile	29	10.1					
		RS-706	Lepomis cyanellus	26	9.1					
		RS-706	Other (11 spp)	18	6.3					
		RS-706	Fundulus olivaceus	17	5.9					
		RS-706	Cottus carolinae	9	3.1					
		RS-706	Semotilus atromaculatus	6	2.1					
Caney Creek	OK	RS-728	Campostoma spp.	527	48.1	2	1.04	0.578	61	Good
		RS-728	Etheostoma spectabile	417	38.0					
		RS-728	Phoxinus erythrogaster	142	13.0					
		RS-728	Other (3 spp)	10	0.9					
		RS-704	Cottus carolinae	304	37.0	4	1.56	0.65	61	Good
		RS-704	Campostoma spp.	214	26.0					
		RS-704	Phoxinus erythrogaster	168	20.4					
		RS-704	Luxilus cardinalis	77	9.4					
		RS-704	Other (6 spp)	31	3.8					
		RS-704	Etheostoma spectabile	28	3.4					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
Evansville Creek	OK	RS-693	Campostoma spp.	304	47.7	4	1.54	0.602	72	Good
		RS-693	Etheostoma spectabile	159	25.0					
		RS-693	Lepomis megalotis	54	8.5					
		RS-693	Lepomis cyanellus	41	6.4					
		RS-693	Luxilus cardinalis	33	5.2					
		RS-693	Noturus exilis	24	3.8					
		RS-693	Other (7 spp)	22	3.5					
Little Lee Creek	OK	RS-10003	Campostoma spp.	105	33.2		2.09	0.753	96	Reference
		RS-10003	Luxilus cardinalis	56	17.7					
		RS-10003	Etheostoma spectabile	36	11.4					
		RS-10003	Lepomis megalotis	31	9.8					
		RS-10003	Etheostoma flabellare	24	7.6					
		RS-10003	Noturus exilis	15	4.7					
		RS-10003	Other (6 spp)	14	4.4					
		RS-10003	Micropterus dolomieu	13	4.1					
		RS-10003	Etheostoma blennioides	8	2.5					
		RS-10003	Lepomis cyanellus	7	2.2					
		RS-10003	Semotilus atromaculatus	7	2.2					
		RS-10004	Lepomis megalotis	181	26.2		2.11	0.656	96	Reference
		RS-10004	Luxilus cardinalis	178	25.7					
		RS-10004	Campostoma spp.	98	14.2					
		RS-10004	Etheostoma flabellare	85	12.3					
		RS-10004	Other (18 spp)	79	11.4					
		RS-10004	Noturus exilis	35	5.1					
		RS-10004	Etheostoma spectabile	19	2.7					
		RS-10004	Etheostoma blennioides	17	2.5					
Park Hill Branch	OK	RS-518	Campostoma spp.	751	77.6	3	1.02	0.369	64	Good
		RS-518	Other (11 spp)	74	7.6					
		RS-518	Etheostoma spectabile	63	6.5					
		RS-518	Cottus carolinae	29	3.0					
		RS-518	Semotilus	27	2.8					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
			atromaculatus							
		RS-518	Etheostoma flabellare	23	2.4					
Peacheater Creek	OK	BS-208	Luxilus cardinalis	47	19.7	4	2.22	0.82	76	Good
		BS-208	Cottus carolinae	46	19.3					
		BS-208	Campostoma spp.	30	12.6					
		BS-208	Semotilus atromaculatus	28	11.8					
		BS-208	Etheostoma flabellare	25	10.5					
		BS-208	Etheostoma spectabile	17	7.1					
		BS-208	Phoxinus erythrogaster	15	6.3					
		BS-208	Nocomis asper	11	4.6					
		BS-208	Etheostoma punctulatum	7	2.9					
		BS-208	Noturus exilis	6	2.5					
		BS-208	Other (5 spp)	6	2.5					
Peavine Creek	OK	RS-657	Phoxinus erythrogaster	258	30.6	3	1.67	0.631	69	Good
		RS-657	Cottus carolinae	252	29.9					
		RS-657	Campostoma spp.	148	17.5					
		RS-657	Etheostoma spectabile	116	13.7					
		RS-657	Other (9 spp)	47	5.6					
		RS-657	Luxilus cardinalis	23	2.7					
Sager Creek	OK	BS-HF04	Etheostoma spectabile	179	53.8	3	1.52	0.613	80	Reference
		BS-HF04	Semotilus atromaculatus	54	16.2					
		BS-HF04	Campostoma spp.	24	7.2					
		BS-HF04	Cottus carolinae	23	6.9					
		BS-HF04	Etheostoma punctulatum	19	5.7					
		BS-HF04	Luxilus cardinalis	19	5.7					
		BS-HF04	Noturus exilis	10	3.0					
		BS-HF04	Other (5 spp)	5	1.5					
Scraper Hollow Creek	OK	RS-667	Phoxinus erythrogaster	45	35.2	3	1.73	0.83	57	Fair
		RS-667	Cottus carolinae	26	20.3					
		RS-667	Semotilus atromaculatus	26	20.3					
		RS-667	Etheostoma spectabile	10	7.8					
		RS-667	Campostoma spp.	7	5.5					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
		RS-667	Lepomis cyanellus	6	4.7					
		RS-667	Noturus exilis	6	4.7					
		RS-667	Other (1 spp)	2	1.6					
Shell Branch	OK	RS-793	Phoxinus erythrogaster	23	20.2	2	1.91	0.917	76	Good
		RS-793	Etheostoma punctulatum	20	17.5					
		RS-793	Etheostoma spectabile	19	16.7					
		RS-793	Cottus carolinae	18	15.8					
		RS-793	Luxilus cardinalis	18	15.8					
		RS-793	Etheostoma flabellare	9	7.9					
		RS-793	Noturus exilis	6	5.3					
		RS-793	Other (1 spp)	1	0.9					
Tahlequah Creek	OK	RS-630	Etheostoma flabellare	290	64.3	4	1.12	0.694	62	Good
		RS-630	Phoxinus erythrogaster	69	15.3					
		RS-630	Campostoma spp.	35	7.8					
		RS-630	Etheostoma spectabile	34	7.5					
		RS-630	Semotilus atromaculatus	23	5.1					
		RS-578	Campostoma spp.	291	40.5	4	1.66	0.556	74	Good
		RS-578	Luxilus cardinalis	230	32.0					
		RS-578	Etheostoma spectabile	67	9.3					
		RS-578	Other (14 spp)	50	7.0					
		RS-578	Noturus exilis	36	5.0					
		RS-578	Cottus carolinae	25	3.5					
		RS-578	Etheostoma flabellare	20	2.8					
Tate Paris Creek	OK	RS-770	Etheostoma flabellare	150	36.2	3	1.69	0.66	78	Good
		RS-770	Etheostoma spectabile	112	27.1					
		RS-770	Luxilus cardinalis	57	13.8					
		RS-770	Campostoma spp.	50	12.1					
		RS-770	Other (7 spp)	21	5.1					
		RS-770	Etheostoma punctulatum	13	3.1					
		RS-770	Semotilus atromaculatus	11	2.7					
Tyner Creek	OK	RS-541	Phoxinus erythrogaster	226	52.3	3	1.01	0.628	53	Fair
		RS-541	Cottus carolinae	157	36.3					
		RS-541	Etheostoma flabellare	42	9.7					
		RS-541	Other (2 spp)	7	1.6					

Creek Name	State	Location	Name	Count	Percent	Stream Order	SW Diversity	SW Evenness	IBI Score	IBI Description
		RS-548	Cottus carolinae	292	47.9	5	1.53	0.614	62	Good
		RS-548	Phoxinus erythrogaster	128	21.0					
		RS-548	Campostoma spp.	86	14.1					
		RS-548	Etheostoma flabellare	41	6.7					
		RS-548	Luxilus cardinalis	26	4.3					
		RS-548	Semotilus atromaculatus	15	2.5					
		RS-548	Other (6 spp)	21	3.4					

SECTION 4
PHOSPHORUS IMPACTS ONLY A SMALL PORTION OF LAKE TENKILLER

4.1 SUMMARY OF DETAILED FINDINGS

- Much of Lake Tenkiller does not "see" the phosphorus that enters from upstream because it plunges to the lake bottom waters.
- Reductions in phosphorus load will only impact the chlorophyll-a levels in the riverine section of the lake.
- Dissolved oxygen depletion at depth is common in deep reservoirs.
- The fisheries in Lake Tenkiller are not damaged.

4.2 MUCH OF LAKE TENKILLER DOES NOT "SEE" THE PHOSPHORUS THAT ENTERS FROM UPSTREAM BECAUSE IT PLUNGES TO THE LAKE BOTTOM WATERS

Longitudinal patterns in reservoirs like Lake Tenkiller determine how the reservoir will respond to pollutants that enter into it (James et al. 1987; Thornton et al. 1990, Chapter 5). These longitudinal gradients result in three distinct zones with unique physical, chemical, and biological characteristics; a riverine zone, a zone of transition and a lacustrine zone (Thornton et al. 1990). The riverine zone is relatively narrow, generally shallow, and well-mixed. Velocities are decreasing, but advective forces are still strong enough to transport finer suspended particles. Primary production in this zone is limited mainly due to limited light penetration. The transition zone is marked by increased light penetration and significant sedimentation (Thornton et al. 1990). The lacustrine zone is analogous to a lake system. Light penetration is sufficient for primary production, which is potentially nutrient-limited in this zone.

Density differences between the water flowing into a reservoir and the water in the reservoir can cause the inflow to plunge and create an underflow or interflow (Thornton et al. 1990). Differences in density can be attributed to temperature, total dissolved solids (TDS) and total suspended solids (TSS), however, temperature is often most important because small changes in temperature have a relatively large affect on density, whereas TDS and TSS concentrations must increase/decrease dramatically to affect density (Thornton et al. 1990). Profiles of temperature, conductivity, dissolved oxygen, and turbidity measured during June-September, 2006 for Lake Tenkiller indicate that the water flowing into the lake likely plunges below the surface in the vicinity of sampling station, LK-04. The plunge point can move due to changing flow conditions (Thornton et al. 1990). Temperature profiles plotted in Figure 4-1 show that the water entering the reservoir, as monitored at river station RS-3, is always colder than the surface water at LK-04. This temperature difference likely causes the inflow to the reservoir to plunge below the surface.

Figure 4-2 shows the locations of the four water quality stations in Lake Tenkiller and the location of the old river channel (thalweg). As seen in this figure, stations LK-03 and LK-01 are outside the thalweg. This fact complicates the flow analysis because the water that plunges beneath the surface can follow the old river channel (thalweg) as an underflow (Thornton et al. 1990) that would not have been sampled at LK-03. Despite this data limitation, plunging is evident from the fact that the water at LK-04 does not "look like" the water sampled at LK-03, -02 or -01.

Turbidity is about 2 times higher at the surface at LK-04 compared to the other three stations and increases sharply (3-5 times) between the surface and bottom (~6 m) of the water column (Figure 4-3). In contrast, the profiles at LK-03, -02, and -01 show relatively low, constant turbidity measurements of <5 Nephelometric Turbidity Units (NTUs) with occasional increases at the very bottom of the water column below about 20 m at LK-01 and LK-02.

Conductivity varies from about 280-340 µS/cm at LK-04 and is consistently higher than at the other three stations, which have conductivities in the range of about 220 to 260 µS/cm (Figure 4-4).

Dissolved oxygen at the surface at Station LK-04 is generally higher compared to the other stations and shows a rapid decline from the surface down to the bottom (4-6 m) where it sometimes reaches 0 mg/L (Figure 4-5). The sampling period August 9, 2006 is an exception where the dissolved oxygen profile for LK-03 is more similar to LK-04. The top 8-10 m of water column at the two deep stations, LK-02 and LK-01 have consistent dissolved oxygen concentrations between about 7 and 9 mg/L with the exception of July 13, 2006 sampling which has the surface down to ~7 m at 10 mg/L.

Profiles of total phosphorus and SRP show that concentrations at LK-04 are generally in the same range as concentrations measured at the river sampling station RS-3 (Figures 4-6 and 4-7). The top 6 meters of water at LK-03 consistently exhibit much lower concentrations of total phosphorus and SRP, as do all depths measured at LK-02 and LK-01 with the occasional exception of the bottom-most sample (or next sample up from bottom in October and November) at LK-02. Unfortunately, because the profiles at LK-03 and LK-01 were not measured in the thalweg, bottom total phosphorus and SRP are not known. What is evident is that the phosphorous concentrations in the water column at LK-04 are similar to concentrations at river station RS-3 and not the same as the top 6 m at the downstream stations in the lake supporting the hypothesis that the inflow is plunging in the vicinity of LK-04.

4.3 REDUCTIONS IN PHOSPHORUS LOAD WILL ONLY IMPACT THE CHLOROPHYLL-A LEVELS IN THE RIVERINE SECTION OF THE LAKE

Of the data collected by the Plaintiffs, the 2006 data provide the best ability to study summer (May to September) conditions Lake Tenkiller. The spatial patterns of 2006 summer average total and soluble reactive phosphorus concentrations in the top two meters are shown in Figure 4-8. Total phosphorus concentration averages about 0.09 mg/L just upstream of the lake (RS-3) and at the riverine station LK-04. A sharp drop to about 0.03 mg/L occurs by the transitional station LK-03, presumably due to the inflow plunging to deeper water. Further drops occur moving to and through the lacustrine section to concentrations between 0.01 and 0.02 mg/L. SRP exhibits a somewhat different pattern. Although total phosphorus does not drop between the upstream river and the riverine station, SRP drops from about 0.065 mg/L to about 0.03 mg/L. The drop is likely due to incorporation of SRP into phytoplankton. By the transitional station LK-03 the SRP has been nearly completely depleted and very little exists in the surface waters of the lacustrine stations.

The pattern in chlorophyll-a is consistent with the pattern in SRP. Between RS-3 and LK-04 there is a substantial increase in chlorophyll-a concentration, reflecting the growth of phytoplankton that occurs over the time it takes water to move between these stations. Consistent with the drop in SRP to very low levels by station LK-03, there is a drop in chlorophyll-a. This decline continues through the lacustrine section where there is too little SRP to support phytoplankton growth. The decline probably represents dilution of the phytoplankton that had grown upstream in the riverine (and perhaps transitional) section.

The data reveal that the phosphorus entering the lake from the river stimulates phytoplankton growth in the riverine section of the lake. SRP is rapidly depleted by phytoplankton uptake and plunging of the inflow below the surface waters where phytoplankton can grow. Little phytoplankton growth occurs in the lacustrine section of the lake. Thus, it is only the riverine and perhaps transitional sections that are impacted by the phosphorus entering the upstream river system from the various sources.

The conditions in the riverine section at LK-04 during 2005 and 2007 are less well defined because of more limited sampling. As shown in Table 4-1, it appears that 2006 was less productive than 2005 and similar to 2007. Note that 2006 exhibited a higher maximum chlorophyll-a than 2007, so perhaps there has been a downward trend in productivity over the three year period.

Table 4-1. Summary of corrected chlorophyll-a concentrations measured in the top 2 meters of riverine section of Lake Tenkiller from May through September of 2005-2007.

Year	Number of Observations	Average	Maximum	Minimum
2005	5	53	133	15
2006	11	22	46	1.2
2007	3	25	26	23

4.4 DISSOLVED OXYGEN DEPLETION AT DEPTH IS COMMON IN DEEP RESERVOIRS

Deep reservoirs typically experience thermal stratification in the summer, which isolates the cooler, deeper portion of the lake (hypolimnion) from the surface waters (epilimnion). In these deeper waters, there is little photosynthesis (because there is little light) and little downward mixing of dissolved oxygen from the epilimnion.[24] Dissolved oxygen concentration declines over the summer due to degradation of organic matter in the water and in the bottom sediments. The decaying organic matter comes from algae that grew upstream or in the epilimnion, wastewater treatment plant discharges, and runoff. The reservoir traps this organic matter, which otherwise would move along with the river, and its decay after settling to the bottom typically depletes the dissolved oxygen. Natural organic matter from forests, wetlands, and fields is an important factor in dissolved oxygen depletion and even oligotrophic reservoirs can experience anoxic conditions in the summer. Figure 4-9 illustrates the hypolimnion dissolved oxygen depletion phenomenon for five Oklahoma reservoirs and compares their dissolved oxygen profiles to that of Lake Tenkiller. All of the lakes in Figure 4-9 are listed as mesotrophic or oligotrophic using the chlorophyll-a Trophic State Index. Although the reservoirs have varying depths, they all experience an anoxic zone. Elmer Thomas Lake, which is assessed as oligotrophic (i.e., the lowest level of eutrophication possible using trophic state), has a low dissolved oxygen layer beginning at four meters in late July 2006, six meters shallower than Lake Tenkiller's low dissolved oxygen layer in July, one year later. In fact, a review of all oligotrophic and mesotrophic lakes in the 2007 BUMP report (OWRB 2007) show that only one did not experience significant depletion of dissolved oxygen in its bottom waters. That one lake, Lake McAlester, is relatively shallow (~9 meters) and most likely does not undergo stratification because of wind-driven mixing in the summer.

[24] As discussed in Section 2.9, during stratification, there is little to no mixing between the epilimnion and hypolimnion. Consequently the oxygen rich upper layers of a deep lake will not mix with the hypolimnion until turnover.

4.5 THE FISHERIES IN LAKE TENKILLER ARE NOT DAMAGED

Lake Tenkiller has catch-limits in place to increase or decrease the catch rate or sizes of different species targeted by anglers. In addition, lake levels are primarily managed for flood control purposes, which can lead to stress or recruitment failure for some species depending on the timing and extremity of water level fluctuations.

The Oklahoma Department of Wildlife Conservation (ODWC) actively manages the bass fishery in Lake Tenkiller, as well as other lakes and reservoirs in the state. The lake has been stocked since inundation with largemouth bass (Florida strain), walleye, striped bass, rainbow trout, threadfin shad (to provide a forage base for bass), and more recently (1990-1991) with smallmouth bass (non-native Tennessee Lake strain) (ODWC 1989, 2003a). Periodic electrofishing surveys are conducted at locations within the riverine, transitional, and lacustrine portions of the lake to assess the bass and other sport fish populations. Based on those studies, Lake Tenkiller typically ranks in the top five in Oklahoma in the number of largemouth bass caught per hour in reservoirs >1,000 acres (ODWC 2003b, 2006). According to ODWC, high quality lakes produce at least 60 bass per hour of electrofishing with 15 or more of those fish at least 14 inches (356 mm) long. Lake Tenkiller was in the high quality category for every year data were available between 1993 and 2006 (Table 4-2; Figure 4-10). In comparison, Broken Bow was below average in 1993-1996 and 2006; in the quality category in 1997; high quality in 2000, 2001.

Table 4-2. Summary of Oklahoma Department of Wildlife Conservation spring largemouth bass electrofishing surveys 1993-2006.[1]

Year	Bass Abundance (#/hour)	Number Bass Over 14 inches per hour	Heaviest Fish (lbs)	Notes
Lake Tenkiller				
1993	119.1	36.7	6.6	
1994	114	42	6.9	
1996	189	75	6.2	
1997	130	48	7.3	
1999	145	79.7	6.0	
2001	110	31	3.4	2000 winterkill of threadfin shad; largemouth bass virus summer of 2000
2002	64	27	4.1	
2003	77.5	21	4.9	
2005	112.3	39.3	5.9	
2006	69	35	4.0	Low lake levels made sampling difficult so numbers may be unnaturally low
Broken Bow				
1993	37.9	13.4	6.4	
1994	33	5	3.8	
1996	27	7	5.2	
1997	55.6	17.6	6.0	
1999	Not Sampled			
2001	72	22	2.6	
2002	72	22	2.6	
2003	Not Sampled			
2005	Not Sampled			
2006	43	8	5.0	

[1] Data downloaded from www.wildlifedepartment.com on June 12, 2008.
High Quality Fishery: 60 or more bass per hour of electrofishing with 15 or more bass at least 14 inches (356 mm) in length.
Quality Fishery: 40 or more bass per hour of electrofishing with 10 or more bass at least 14 inches (356 mm) in length.

Lake Tenkiller has been cited as one of the "state's premier fisheries" with fishing for black bass, crappie, and catfish (McNeff 2008). The black bass (i.e., smallmouth, largemouth, and spotted bass) fishery in Lake Tenkiller is dominated by largemouth bass, with smaller numbers of spotted bass and smallmouth bass (Table 4-3).[25] Largemouth bass typically prefer

[25] Note: not all data used in this analysis were provided from the Plaintiffs' laboratory sheets. Additional data were used from Stevenson's considered materials; specifically: "Database CDM 20080518.mdb."

warmer, quieter waters of lakes and large streams compared to smallmouth bass and spotted bass. The large number of coves and backwater areas with vegetative growth are most suitable to largemouth bass. Spotted bass may do well in some clear lakes; however, they are best adapted for small, clear, spring-fed streams (Miller and Robison 2004). Smallmouth bass also prefer cool, clear rocky streams with spawning occurring in flowing waters (Miller and Robison 2004).

In 1987, ODWC modified the black bass (largemouth, smallmouth, spotted bass) fishing catch-limits due to several years of successful recruitment. The limit was changed from a 14 inch (356 mm) minimum size to a slot limit of 13 to 16 inches (330 to 406 mm), with a creel limit of 6 fish per day (combined) above or below this size range. This is typically done in lakes to encourage anglers to harvest the smaller fish that are competing for available forage, essentially thinning out the population so that the remaining fish can grow larger, faster. The slot limit can be used on lakes with numerous years of successful recruitment and an abundance of juveniles. In Lake Tenkiller, bass have much more reproductive success when spring lake levels are in the flood pool, particularly during spawning and rearing (May 15 – July 1). In years when water levels are lower, black bass recruitment is expected to be much less. The catch-limits were changed again in 1997 for spotted bass with no minimum length limit and a creel limit of 15 fish per day, to encourage harvest of this species (ODWC 2003a). In 1997, the smallmouth and largemouth bass limits were not changed. In 2009, catch and size limits were eliminated for spotted bass statewide; limits were unchanged for smallmouth and largemouth bass (ODWC 2009a)

Based on electrofishing data from 1991 through 1997 (ODWC), black bass condition has generally been healthy (i.e., condition factor greater than 1.0; Table 4-3). Largemouth bass numbers declined slightly lake wide from 1991 to 1997, with spotted bass slightly increasing over that time frame and smallmouth bass remaining low (Table 4-3).

Table 4-3. Lake Tenkiller spring (April-May) electrofishing sampling - bass condition factor.

Zone	Year	Largemouth Bass		Smallmouth Bass		Spotted Bass	
		Condition Factor	Count	Condition Factor	Count	Condition Factor	Count
Riverine	1991	1.25	319	1.12	2	1.28	17
	1992	1.24	352	1.21	1	1.27	29
	1993	1.28	187	0.98	1	1.22	12
	1994	Not Sampled					
	1996	1.32	125	*	*	1.27	9
	1997	1.31	174	*	*	1.36	10
Transitional	1991	1.26	353	*	*	1.33	55
	1992	1.31	358	1.12	2	1.37	52
	1993	1.30	280	*	*	1.26	51
	1994	1.25	254	1.27	1	1.20	52
	1996	1.28	143	*	*	1.30	11
	1997	1.33	185	1.36	3	1.30	57
Lacustrine	1991	1.30	156	1.19	1	1.27	14
	1992	1.25	149	1.33	1	1.11	24
	1993	1.23	51	*	*	1.12	4
	1994	Not Sampled					
	1996	1.36	297	*	*	1.35	18
	1997	1.23	202	1.34	1	1.11	45
Lake wide	1991	1.27	828	1.14	3	1.31	86
	1992	1.27	859	1.20	4	1.28	105
	1993	1.29	518	0.98	1	1.24	67
	1994	1.25	254	1.27	1	1.20	52
	1996	1.33	565	*	*	1.31	38
	1997	1.29	561	1.36	4	1.23	112

** Species not captured during year.*
Data from Oklahoma Department of Wildlife Conservation.
Condition factor = (Wt x 100,000)/(TL^3)

Based on the large data set available for largemouth bass and the management focus on this species, additional analyses were conducted to assess the overall health of this fishery based on length frequency plots and condition factor from the 1991-1997 ODWC dataset. On a lake wide basis, largemouth bass mean length and weight in spring increased from 1991 to 1997 (Table 4-4). This may be the result of the slot limit placed on black bass in 1987 (Smith 1988).

Table 4-4. Lake Tenkiller spring (April-May) electrofishing sampling - largemouth bass.

Zone	Year	Mean TL (mm)	S.D. TL	Max TL (mm)	Mean Wt (g)	S.D. Wt		Mean Condition Factor	Count
Riverine	1991	271	85	560	363	431	3147	1.25	319
	1992	276	83	530	349	348	2041	1.24	352
	1993	292	97	520	462	445	2381	1.28	187
	1994	Not Sampled							
	1996	292	96	520	448	446	2080	1.31	126
	1997	321	93	572	559	504	2940	1.28	178
Transitional	1991	271	85	560	363	431	3147	1.25	319
	1992	276	83	530	349	348	2041	1.24	352
	1993	292	97	520	462	445	2381	1.28	187
	1994	Not Sampled							
	1996	292	96	520	448	446	2080	1.31	126
	1997	321	93	572	559	504	2940	1.28	178
Lacustrine	1991	264	79	510	326	353	2041	1.30	156
	1992	293	82	535	424	305	1701	1.07	174
	1993	303	77	545	425	363	2155	1.23	51
	1994	Not Sampled							
	1996	311	98	556	544	483	2620	1.36	297
	1997	300	104	920	481	905	12000	1.22	203
Lakewide	1991	267	82	560	340	390	3147	1.27	828
	1992	286	82	560	383	335	2381	1.23	884
	1993	294	97	545	469	458	2807	1.29	518
	1994	304	93	750	472	561	6237	1.25	254
	1996	297	97	556	478	456	2620	1.33	566
	1997	314	97	920	536	669	12000	1.27	567

Note: Records with negative weight values were removed from the analysis.
Data from Oklahoma Department of Wildlife Conservation.
TL = total length; S.D. = standard deviation; Wt. = weight
Condition factor = (Wt x 100,000)/(TL^3)

An evaluation of length frequency provides an assessment of overall size structure in the lake. There is little difference in length frequency in largemouth bass among the three zones; therefore, length frequency was evaluated on a lake wide basis to provide a more robust analysis. From 1991 to 1997, it is apparent that the overall size structure has moved towards a more balanced population with more fish in the larger size classes in later years (Figure 4-11). As noted above, the largemouth bass fishery declined in overall numbers from 1991-1997, but through management of the fishery the remaining fish are larger and more evenly distributed, which was a focus of the lake management and implementation of slot limits.

Two additional metrics, commonly used in fisheries management to evaluate size structure are the proportional stock density (PSD) and relative stock density (RSD). The proportional stock density is calculated as:

$$\mathrm{PSD} = \frac{\text{number of fish} \geq \text{minimum quality length}}{\text{number of fish} \geq \text{minimum stock length}} \times 100$$

The relative stock density is calculated as:

$$\mathrm{RSD} = \frac{\text{number of fish} \geq \text{specified length}}{\text{number of fish} \geq \text{minimum stock length}} \times 100$$

Both PSD and RSD can range from 0 to 100 and are typically reported to the nearest whole number. Stock length has been defined as the approximate length at maturity, the minimum length effectively sampled by traditional fisheries gears, and the minimum length of fish that provide recreational value (Anderson and Neumann 1996). Quality length is defined as the minimum size fish most anglers like to catch. For largemouth bass, stock length is 8 in. (200 mm) and quality length is 12 in. (300 mm; Anderson and Neumann 1996). RSD was first used for largemouth bass with a specified length of 15 inches (380 mm) which represents the preferred minimum size and is referred to as RSD15 (Anderson and Neumann 1996). Two other size ranges can be used in the RSD, memorable minimum size and trophy minimum size. Based on data from the 1987 assessment, PSD was calculated as 60 and RSD15 was 17 for largemouth bass (ODWC 1989). In 1997, PSD was 66 and RSD15 was 31. The change in regulations in 1987 appeared to result in are higher proportion of larger fish by 1997 as intended. Balanced largemouth bass populations may have PSD values ranging from 40 to 70 (Anderson and Neumann 1996).

The largemouth bass population in Lake Tenkiller is the most abundant of the black bass; spotted bass makes up approximately 20% of the black bass population. The goal of management is to maintain the spotted bass population at 15% to 20% of the total black bass population (Smith 1988). Catch-limits were modified for spotted bass in 1997 with no minimum length limit and a creel limit of 15 fish per day to encourage more angling for this species (ODWC 2003a). Lake Tenkiller was recently included on a list of the Best Fishing Towns in America by *Field and Stream* (Deeter 2008). Tahlequah, Oklahoma was the focus with reference to the proximity of the town to prime largemouth bass fishing lakes, especially Lake Tenkiller.

A largemouth bass die off occurred in Lake Tenkiller in the summer of 2000 due to an outbreak of largemouth bass virus (ODWC 2003a). All largemouth bass tested in 2000 were infected with the virus while these numbers dropped in subsequent years with just over 11% of the tested fish infected in 2003 (ODWC 2003a). This may have reduced the population slightly, but it appears to be recovering. In addition, a winterkill of threadfin shad in winter 2000 resulted in a smaller forage base available for bass the following spring. This may have resulted in the slightly lower condition of bass as reported by ODWC (2003a).

While smallmouth bass may be a desirable species in the Lake Tenkiller black bass fishery, it is likely that the endemic smallmouth bass (Micropterus dolomieui velox) in the Illinois River Watershed is adapted for more of a riverine environment. The Neosho strain (*M. dolomieu velox*) of smallmouth bass present in the Illinois watershed represents one of the most isolated populations of the species (e.g., high genetic diversity; Oklahoma State University [OSU] 1994). Prior to the formation of Lake Tenkiller, spotted bass was the dominant bass within this section of the Illinois River (Paden 1948), with smallmouth bass in moderate numbers and largemouth bass least dominant. Following formation of Lake Tenkiller, the status of the smallmouth bass population in the lake was in doubt due to their habitat requirements (Hall 1953). Twenty five years later, a regional fisheries biologist indicated that there were no lakes in Oklahoma that he knew of where spotted bass adapted more successfully than largemouth bass when impounded (Smith 1978). Additionally, spotted bass generally adapt better than the smallmouth bass to impoundment conditions, but the largemouth bass are dominant (Smith 1978). Therefore, one would expect the bass fishery in Lake Tenkiller to be dominated by largemouth bass, followed by spotted bass, with smallmouth bass a minor component. Based on management reports through the 1980s, black bass management in Lake Tenkiller focused primarily on largemouth bass as the dominant black bass in the system.

Smallmouth bass of a non-native reservoir strain (Tennessee Lake strain) were stocked in 1990 and 1991 in an effort to develop a more robust smallmouth bass fishery based on a more lacustrine adapted strain. This effort was suspended following analysis of the genetic diversity in the native strain smallmouth bass (OSU 1994). This suspension was based on a study of the genetic distinctiveness of smallmouth bass in Oklahoma that found "..the Neosho and Ouachita forms of smallmouth bass are the most isolated populations of the species…and protein electrophorsis demonstrates that, genetically, the Neosho and Ouachita forms of smallmouth bass are the most distinctive of all populations of smallmouth bass….likely the result of a long history of isolation that probably dates to the last glaciation (10,000 years ago) or earlier" (OSU 1994). Based on those findings the authors recommended "…no stockings of non-native smallmouth bass in the Little, Neosho, and Illinois river systems or direct tributaries of the Arkansas River" (OSU 1994).

The assessment by Drs. Cooke and Welch on the habitat squeeze for smallmouth bass and spotted bass within Lake Tenkiller relies on a gross approximation of general habitat conditions within the main channel of the lake. Application of this throughout the entire lake (based on four open water sampling locations) to assess habitat availability for black bass does not account for life history strategies of these species. Black bass are a littoral zone species, occupying steep rocky shorelines or areas with macrophyte coverage, while the habitat squeeze model is based on water quality within the pelagic zone. The habitat squeeze model does not represent or account for the numerous refuges available within the littoral zone, especially at the mouths of tributaries and in coves. Smallmouth bass in lakes and reservoirs typically prefer drop-offs, rocky shoals, and wave swept littoral regions (Edwards et al. 1973; Hubert and Lackey 1980; Winemiller and Taylor 1987). Spotted bass also prefer areas with steep, rocky shorelines (McMahon et al. 1984). Adult black bass typically feed in the littoral zone, with smallmouth bass and spotted bass feeding on crustaceans and fish within the interstitial spaces in cobble and largemouth bass feeding primarily on prey found within vegetated habitats (Werner et al. 1977; McMahon et al. 1984; Weaver et al. 1997).

The morphology of Lake Tenkiller, with steep, rocky shoreline areas provides the necessary littoral zone habitat for smallmouth bass that would not be compromised by the open water hypoxia. The littoral zone in these areas is not likely to experience the oxygen depletion seen in the hypolimnion due to wind and wave action and fresh water inflow from tributaries. The presence of a healthy population of spotted bass (catch limits are currently set to encourage harvest of this species because their population is higher than fishery managers prefer) also indicates that there are refuges available for this species throughout the lake during the warmest summer months. An evaluation of littoral zone habitat availability and suitability, including temperature and dissolved oxygen fluctuations, would provide more suitable information to describe factors that may be influencing black bass populations within Lake Tenkiller.

In addition to black bass, several other sportfish are found in Lake Tenkiller including white bass and channel catfish. Channel catfish and white bass populations were assessed based on gillnetting surveys conducted by ODWC during November 1990, 1992, 1993, and 1996. Overall condition of both species was good with condition factors above 1.0 each year (Table 4-5). Based on the length frequency analysis, many of the white bass collected through the years are within the preferred size (minimum length 12 in; 300 mm) for anglers (Figure 4-12). For white bass, the preferred size range is 12 inches (300 mm) or greater total length, with memorable size 15 inches (380 mm) or greater, and trophy 18 inches (460 mm) or greater. Approximately 8% of the samples in 1991 were of trophy size, with lower percentages in subsequent years (Figure 4-12). Overall abundance of white bass was similar in all years to the range seen from 1978 to 1987 (113 to 265 individuals) (ODWC 1989).

Table 4-5. Lake Tenkiller November gillnet and research gillnet sampling summary.

Zone	Year	Mean TL (mm)	TL S.D.	Max TL (mm)	Mean Wt (g)	Wt S.D.	Max Wt (g)	Mean Condition Factor	Count*
Channel Catfish									
Riverine	1990	409	199	1050	873	925	4082	1.16	59
	1992	241	102	650	436	635	3990	1.00	119 (62)
	1993	268	108	525	354	447	2098	1.21	38
	1996	Not Sampled							
Transitional	1990	273	132	635	463	855	3402	0.96	19
	1992	291	75	414	353	298	940	1.11	24 (17)
	1993	224	86	550	195	331	2041	1.07	52
	1996	269	88	487	359	473	2040	1.06	27
Lacustrine	1990	356	130	765	832	1150	5897	1.16	28
	1992	243	88	420	550	452	1276	1.38	22 (9)
	1993	286	110	590	443	520	2835	1.25	52
	1996	300	94	600	429	523	2722	1.17	25
Lakewide	1990	371	179	1050	789	981	5897	1.13	106
	1992	249	98	650	432	566	3990	1.06	165 (88)
	1993	258	104	590	328	449	2835	1.17	142
	1996	284	91	600	393	494	2722	1.12	52
White Bass									
Riverine	1990	299	57	369	252	126	400	0.87	11
	1992	266	91	503	463	408	1644	1.10	46 (27)
	1993	234	87	500	191	232	1216	1.22	79
	1996	Not Sampled							
Transitional	1990	282	69	440	363	243	1021	1.36	18
	1992	315	135	630	820	840	2892	1.29	24 (17)
	1993	224	72	510	169	220	1382	1.10	228
	1996	286	93	550	422	411	2800	1.25	133
Lacustrine	1990	314	121	690	671	1135	6208	1.31	62
	1992	300	91	476	662	491	1758	1.35	50 (29)
	1993	263	103	670	355	513	4536	1.25	433
	1996	262	82	450	276	273	1042	1.05	132
Lakewide	1990	306	106	690	560	956	6208	1.27	91
	1992	290	102	630	625	574	2892	1.25	120 (73)
	1993	248	94	670	280	427	4536	1.20	740
	1996	274	88	550	349	356	2800	1.15	265

**Data from 1992 contained length data for all fish, but was missing weight data for a subset; weight count provided in parentheses. Data from Oklahoma Department of Wildlife Conservation.*
TL = total length; S.D. = standard deviation; Wt. = weight
Condition factor = (Wt x 100,000)/(TL3)

For channel catfish, many of the fish captured during the gillnetting surveys were within the stock (11 inches; 280 mm minimum total length) or quality size range (16 inches; 410 mm minimum total length) (Figure 4-13). In 1990, nearly 10% of the individuals were near memorable size (minimum length 28 inches; 710 mm), while in subsequent years, most were in the quality size class (16 inches; 410 mm) with a few in the preferred size class (24 inches; 610 mm) in subsequent years. This is a slight shift from 1987 when 42% were within the quality size class, indicating an overall smaller fish and likely younger population. There is no size limit on channel catfish within Lake Tenkiller with a maximum limit of 15 (channel and blue catfish combined) fish per day. Overall abundance is generally within the range reported from 1978 to 1987 (45 to 121 individuals) (ODWC 1989), and although being low in some years and slightly smaller in median length, is relatively balanced.

An overall assessment of species relative abundance based on a combination of seining data and gill net data also was conducted based on ODWC data. All species captured during the fall gillnetting in 1990, 1992, 1993, and 1996 were recorded and provide an overall assessment of species composition within the three zones of Lake Tenkiller (Riverine, Transitional, and Lacustrine). Zones were defined as provided in Cooke and Welch's May 2008 report. It is interesting to note, that from 1990 to 1996, white bass percent frequency from samples within the three zones increased from 10% to 30% to approximately 30% to 60% of the composition in 1993 and 1996 (Figure 4-14). White bass feed on threadfin shad which were stocked in Lake Tenkiller since as early as 1965 (ODWC 1989). In addition, gizzard shad numbers have been fairly high during this time period as well (Figure 4-14). Threadfin shad typically provide better forage of the two shad species, since they do not get as large and grow slower than gizzard shad, thereby spending more time as available prey. This increased forage base may have allowed for an increase in the white bass population.

Analysis of seining data obtained for June and July of 1990 and 1991 provides insight into the forage base and recruitment of many species. As would be expected, some species are abundant within one zone and not another and catches were typically dominated by a few species. Brook silversides was most abundant in both years in the transitional and lacustrine zones while shad were dominant in the riverine zone both years (Table 4-6 and Figure 4-15).

Table 4-6. Summary of seining data during June and July 1990 and 1991.

Species	Riverine		Transitional		Lacustrine	
	Count	Percent	Count	Percent	Count	Percent
1990						
shad	8198	81.88	464	19.17	0	0.00
brook silverside	401	4.01	614	25.36	1523	68.95
sunfish	391	3.91	83	3.43	90	4.07
central stoneroller	369	3.69	53	2.19	76	3.44
shiner	362	3.62	366	15.12	163	7.38
minnow	163	1.63	55	2.27	35	1.58
largemouth bass	46	0.46	118	4.87	78	3.53
smallmouth buffalo	31	0.31	0	0.00	24	1.09
bluegill sunfish	27	0.27	44	1.82	9	0.41
western mosquitofish	9	0.09	1	0.04	4	0.18
channel catfish	3	0.03	27	1.12	0	0.00
white bass	3	0.03	53	2.19	18	0.81
darter	2	0.02	0	0.00	1	0.05
white crappie	2	0.02	2	0.08	0	0.00
buffalo	1	0.01	0	0.00	0	0.00
gizzard shad	1	0.01	2	0.08	6	0.27
green sunfish	1	0.01	2	0.08	3	0.14
smallmouth bass	1	0.01	1	0.04	3	0.14
walleye	1	0.01	1	0.04	0	0.00
chub	0	0.00	0	0.00	0	0.00
freshwater drum	0	0.00	4	0.17	0	0.00
logperch	0	0.00	2	0.08	1	0.05
longear sunfish	0	0.00	0	0.00	2	0.09
river carpsucker	0	0.00	0	0.00	0	0.00
spotted bass	0	0.00	11	0.45	18	0.81
threadfin shad	0	0.00	518	21.40	155	7.02
1991						
shad	23181	91.19	672	28.19	274	15.58
brook silverside	1235	4.86	1144	47.99	980	55.71
shiner	327	1.29	121	5.08	179	10.18
minnow	324	1.27	198	8.31	86	4.89
central stoneroller	118	0.46	103	4.32	43	2.44
white crappie	90	0.35	0	0.00	0	0.00
gizzard shad	41	0.16	0	0.00	30	1.71
sunfish	38	0.15	25	1.05	83	4.72
western mosquitofish	31	0.12	0	0.00	0	0.00
bluegill sunfish	9	0.04	38	1.59	2	0.11
spotted bass	9	0.04	9	0.38	46	2.62
largemouth bass	5	0.02	19	0.80	9	0.51
river carpsucker	5	0.02	0	0.00	0	0.00
logperch	3	0.01	4	0.17	1	0.06
chub	2	0.01	0	0.00	0	0.00
threadfin shad	2	0.01	28	1.17	14	0.80

Species	Riverine		Transitional		Lacustrine	
	Count	Percent	Count	Percent	Count	Percent
buffalo	0	0.00	0	0.00	0	0.00
channel catfish	0	0.00	10	0.42	0	0.00
darter	0	0.00	0	0.00	0	0.00
freshwater drum	0	0.00	0	0.00	0	0.00
green sunfish	0	0.00	9	0.38	7	0.40
longear sunfish	0	0.00	3	0.13	0	0.00
smallmouth bass	0	0.00	0	0.00	5	0.28
smallmouth buffalo	0	0.00	0	0.00	0	0.00
walleye	0	0.00	0	0.00	0	0.00
white bass	0	0.00	1	0.04	0	0.00

Data from Oklahoma Department of Wildlife Conservation.
Data are sorted in descending order according to species percentage in the riverine section.

While one dataset alone cannot provide an overall assessment of the health of the Lake Tenkiller fishery, an assessment of various datasets including sportfish collections from electroshocking and gillnet sets and forage and recruitment from seining data can assist in understanding fisheries dynamics within the lake. Differences may be expected due to the three zones observed in Lake Tenkiller (Riverine, Transitional, Lacustrine) and species requirements. Assessing these at both the lake wide and zone level have provided insight into the Lake Tenkiller fisheries. This lake is a managed fishery with catch-limits put in place to increase the catch rate or sizes of fish captured by anglers. In addition, lake levels also are fairly strictly managed, which can lead to stress or recruitment failure for some species depending on the timing and extremity of water level fluctuations. Catch rates for bass are among some of the highest in the state with most years qualifying as a high quality bass fishery. Largemouth bass are the most highly sought bass species for anglers, and management strategies have focused on creating a quality largemouth bass fishery, including development of a Largemouth Bass Management Plan by the ODWC (ODWC 2009b). Based on the discussion and analyses above, it apparent that Lake Tenkiller is a desirable fishery. The lake fishery is in no way "damaged," as reported by the Plaintiffs' consultants, Drs. Cooke and Welch.

SECTION 5
BACTERIA SOURCES CAUSE LITTLE RISK OF GASTROINTESTINAL ILLNESS FOR RECREATIONAL USERS OF THE ILLINOIS RIVER IN OKLAHOMA

In his May 2008 report, Dr. Fisher states "Poultry are highly significant contributors to bacterial pollution of surface and ground water within the Illinois River Watershed". Bacterial contamination is discussed in more detail in the May 2008 reports submitted by Drs. Teaf and Harwood. This section examines the validity of some claims made in these reports regarding the degree of bacterial impacts in the Illinois River Watershed, the amount of those impacts attributable to poultry, and the rate of illness that can be expected.

5.1 SUMMARY OF DETAILED FINDINGS

- Dr. Teaf's description of the relationship between indicator bacteria density and illness is incorrect.
- Indicator bacteria loading estimates to the Illinois River Watershed overstate the poultry contribution.
- The Plaintiffs' consultants misrepresent problems known with indicator bacteria.

5.2 A REVIEW OF RECREATIONAL WATER QUALITY STANDARDS FOR BACTERIA INDICATORS

Illness-causing microorganisms (pathogens) that may be contracted by humans during recreational use of surface waters include a wide range of bacteria, viruses, and protists. Low-cost, high-throughput methods to test for the multiple strains and species of common pathogens are still many years from implementation (CIESM 2004; USEPA 2007a, 2007b; Wade et al. 2008). The USEPA therefore established water quality criteria for three bacteria groups that may be found in human feces and that can be easily cultured. These bacteria groups may all have non-human sources in the environment, including animal feces, which have an undetermined disease potential. Therefore their use as an indicator of illness risk has been widely questioned (USEPA 1986, 2007a; National Research Council [NRC] 2004). Density of these indicator bacteria groups is nonetheless regulated as an indirect way of reducing risk of developing illness as a result of full body contact recreational use of water. Water quality criteria for these indicator bacteria were developed primarily as guidelines for beach closures. They can also be applied in cases of less regular or complete body to water contact, such as a capsize event during floating, although the risk of illness may be reduced in these situations.

The USEPA bacterial water quality criteria, and the data used to develop them, have not changed since 1986. The standards for *Escherichia coli (E. coli)* and enterococci are based upon epidemiological data collected by the USEPA in the 1970s and 1980s (USEPA 1986). Current fecal coliform criteria originate from relatively poor epidemiological data gathered in the 1940s and 1950s. Because its later studies showed no correlation between illness and fecal coliform incidence, the USEPA has since 1986 recommended discontinuing the use of fecal coliform water quality standards (USEPA 1986, 2001a).

Individual states enforce standards for one or more of the three indicator groups, usually at or near the level recommended by the USEPA. In spite of USEPA recommendations to discontinue the use of fecal coliform standards (USEPA 1986, 2001a), fecal coliform is still the most commonly enforced bacterial water quality standard in the United States, likely because of its ease of use (USEPA 2001a; PBSJ 2000). The state of Oklahoma is unusual in enforcing *E. coli* and enterococci standards in addition to a fecal coliform standard. Arkansas only enforces a fecal coliform standard.

5.3 TEAF'S DESCRIPTION OF THE RELATIONSHIP BETWEEN INDICATOR BACTERIA DENSITY AND ILLNESS IS INCORRECT

In his May 2008 report, Dr. Teaf presents Figure B2 (Figure 5-1 in this report). The figure is a reproduction of a conceptual model from USEPA (2004) of the relationship between water quality indicator bacteria density and swimming-associated illness rates. Dr. Teaf incorrectly describes this conceptual model in a way that exaggerates the apparent illness risk from indicator bacteria exceedances. The misleading description consists of two main errors:

- Dr. Teaf states that illness rates resulting from swimming increase exponentially at the point where USEPA water quality thresholds for indicator bacteria are exceeded, but the USEPA conceptual model shows the exponential increase does not occur until higher bacteria levels.
- Dr. Teaf claims that this model is based upon data, but there are at this time no epidemiological data to support the USEPA's conceptual model and a large body of data refutes this model.

The following quotes (emphasis added) are from Dr. Teaf's May 2008 report (paragraphs 22 and 23)

For E. coli, a geometric mean density of 126 organisms per 100 milliliters (ml) of water over a 30-day period was associated with an illness rate of 0.876, or 8 illnesses per 1,000 recreational users. As a short-term measure, this 0.8% illness rate was associated with bacterial counts of 236 per 100 ml as an upper limit. For the enterococci, a geometric mean of 33 organisms per 100 ml and an upper limit of 62 organisms per 100 ml were associated with the 0.8% illness rate (OAC, 2007). ***Above these thresholds the agency noted that illness rates rise sharply*** *and the health-based recommendation seeks to remain below that part of the statistical curve (Figure B2).*

and:

While states can determine their own disease incidence targets, the EPA (2003) recommends less than 1.0% illness rate (e.g., 8% is used by many states). This is in part because ***the epidemiological data illustrate an exponential increase in rates beyond the 1.0% illness incidence threshold, instead of a linear increase.*** *Stated differently, at low indicator organism densities (i.e. below the health-based recommendation) the pattern of increase in illness is fairly small and the line is flat, while at higher indicator organisms [sic] densities, the illness rate curve rises much more steeply (Figure B2).*

The above statements by Dr. Teaf are incorrect in part because the inflection point of the USEPA conceptual model is not at the water quality threshold, but at a higher level – a level beyond the extent of the USEPA epidemiological data. These statements by Dr. Teaf are also incorrect in that the change to a steeper illness rate curve is purely hypothetical on the part of the USEPA, not based upon data. In fact, the USEPA (USEPA 2004) provides no explanation of why an exponentially increasing dose response curve is expected. Their actual data "fit linear regression models well, and are considered to characterize the initial 'flat' portion of the conceptual dose-response relationship" (USEPA 2004, Figure 5-2).

The fresh water epidemiological study data, which the USEPA used to set the current threshold criteria for *E. coli* and enterococci (USEPA 1986), are reproduced here in Figure 5-2. As stated by the USEPA, these data show no increase in illness rate vs. bacteria count at the water quality standard threshold. In fact, the USEPA data show that illness rates increase at a much less than 1:1 ratio vs. increasing bacteria count. Least squares linear regressions of the data (Figure 5-2) show that one additional illness per 1000 swimmers is predicted when the water quality thresholds are exceeded by an additional 20 bacteria per 100 ml of water for *E. coli* (R^2 = 0.77) and 5 bacteria per 100 ml for enterococci (R^2 = 0.64). A Spearman non-parametric linear correlation test was also performed in USEPA 1986. In that statistical test, correlation coefficients lower than 1 indicate a positive, but less than 1:1 rate of change with bacteria numbers, and the USEPA found correlation coefficients of 0.80 and 0.74 for *E. coli* and enterococci respectively vs. illness rates in bathers (USEPA 1986).

Reviews and meta-analyses of data from multiple freshwater epidemiological studies also fail to support the USEPA conceptual model of an exponentially increasing dose response curve and demonstrate that illness rates increase at a less than 1:1 rate vs. indicator bacteria density (Pruss 1998; Wade et al. 2003). Pruss (1998), in an analysis of seven freshwater epidemiological studies, found that rates of illnesses actually increase at a declining rate with increasing indicator bacteria density (Figure 5-3). Wade et al. (2003), in a review of 10 freshwater epidemiological studies concluded "a log (base 10) unit increase in *E. coli* was associated with a 2.12 increase in relative risk in freshwater", meaning a 10-fold increase in indicator bacteria caused only a 2-fold increase in risk of illness. Most recently, Wade et al. (2008), conducted an epidemiological study of Great Lakes beaches using a highly robust experimental design and statistical analysis to compare swimming related illness rates to enterococcus density. Enterococcus was measured by the standard cultural technique and by a molecular genetic technique (quantitative polymerase chain reaction; qPCR) that is currently favored to replace standard cultural techniques. Both techniques again supported a less than 1:1 increase in illness risk vs. indicator bacteria density.

In summary, contrary to the contention of Teaf, the USEPA conceptual model of an exponentially increasing dose response curve does not put the inflection point at the water quality thresholds, but above those thresholds, and, more importantly, an exponential increase has never been substantiated by data. There is currently no scientific reason to think that rate of illness rises more sharply as indicator bacteria density increases, and substantial data to support a less than 1:1 rate of illness increase with increasing indicator bacteria densities.

5.4 INDICATOR BACTERIA LOADING TO THE ILLINOIS RIVER WATERSHED OVERESTIMATES POULTRY CONTRIBUTION

Dr. Teaf attempted to estimate the relative contributions of various sources to the fecal coliform load into the Illinois River Watershed. A main conclusion of this work is that poultry contribute less fecal coliform than cattle, but poultry account for about 41% of the total contribution from all livestock in the watershed (Clay 2008). Although Dr. Teaf concludes that livestock account for nearly all of the fecal coliform loading, he failed to do a fair comparison between inputs from livestock and human and he neglected the contribution from recreational users. As a result, the loading estimate is biased in a way that increases estimated input from poultry with respect to other sources.

5.4.1 Unclear and Unequal Treatment of Sources in Loading Estimates

Dr. Teaf provides estimates of fecal coliform inputs to the Illinois River from both Arkansas and Oklahoma counties in the watershed. The sources included were:

> *…fecal coliform contributions by a variety of categories for which data were available, including: domestic pets, deer/wildlife, failing septic systems, permitted point sources (i.e., NPDES outfalls), and livestock. The livestock category was further subdivided into groups by poultry, cattle/calves, horses/ponies, sheep/lambs, and swine.* (Teaf 2008)

Data sources and methods for estimating livestock inputs were detailed in Table B5 of Dr. Teaf's May 2008 report, but data sources and loading estimate methods were not stated for the failing septic systems or the permitted point sources. The National Pollutant Discharge Elimination System (NPDES) point sources are probably outfalls of WWTPs, but this is never stated. Further, Table B3 of the report confusingly lists only WWTPs from two Oklahoma counties in spite of its title "NPDES facilities in the IRW". Dr. Teaf also does not disclose important details regarding his estimates of septic system and permitted point source loadings. The data used for permitted point sources might be maximum permitted levels for WWTPs or real "end of pipe" data for some undisclosed time period, or something else entirely. Teaf claims that his source analysis was "conducted in a fashion consistent with that employed by USEPA and ODEQ" (Teaf 2008), but the paucity, and in some cases lack, of information provided about Teaf's data sources is in clear violation of Total Maximum Daily Load (TMDL) guidance provided by both the state of Oklahoma (ODEQ undated) and the USEPA (USEPA 2001a). With information about methods and sources of human input estimates simply not available in Dr. Teaf's report, an objective evaluation of the loading estimates is impossible.

The information Dr. Teaf provides about his estimates of fecal coliform loads from livestock indicate he likely biased the estimated fecal coliform load contributed by livestock, including poultry, over the load contributed by humans in the Illinois River Watershed. Livestock loads are based on estimated fecal coliform density in fresh, solid animal waste (Tables B4 and B5 in Dr. Teaf's May 2008 report), making no allowances for die-off due to drying, fermentation, exposure to sunlight, adsorption to soil particles, and dilution of the bacteria before they enter waters used for full body recreational purposes (Clay 2008). The equivalent treatment for human waste would have been to estimate the fecal coliform density in fresh, untreated human feces and to assume this load was deposited directly into the river rather than first passing through septic tanks and ground water or through WWTPs.

5.4.2 Failure to Consider Pollution Input From Human Recreation

Dr. Teaf made no attempt to estimate fecal coliform input to the Illinois River Watershed resulting from human recreational use of the watershed. On average, over 117,000 people per year took float trips down the Illinois River from 2004 to 2007, with a typical single day float trip comprising about four hours of river contact time, and an additional average of almost 40,000 people visit the river for, fishing, camping, or swimming trips every year (Caneday 2008). An average of over 2.6 million campers, boaters and other types of recreational visitors utilize Lake Tenkiller every year (Caneday 2008).

Human recreational use is well known to contribute human fecal bacteria to recreational waters (see Jarman [2008] for specific discussion related to the Illinois River Watershed), which is why the USEPA recommends that beach waters be monitored frequently for indicator bacteria during peak swimming times (USEPA 1986). In addition, there is evidence that sediments under popular swimming locations can build up a reservoir of fecal indicator bacteria that can then serve as a source of these bacteria to the water column (i.e., Christensen et al. 1979; Crabill et al. 1999; Wheeler Alm et al. 2003). Given the high human use of the Illinois River Watershed for recreation, it is not appropriate to construct a loading budget for indicator bacteria (or for other contaminants such as phosphorus and nitrogen) without accounting for human recreational use among the inputs. A budget that does not account for this source overestimates the proportional input from non-human sources.

5.4.3 Use of the Least Appropriate Indicator for Loading Estimates

It is not clear why Dr. Teaf performed his loading estimate analysis for fecal coliform bacteria. Fecal coliforms are well known to be the least reliable of the bacterial indicator groups, with no significant relation to human disease incidence in most epidemiological studies, whereas multiple studies have found *E. coli* correlates best to illness in bathers, followed by enterococci

(USEPA 1986; Pruss 1988; Wade et al. 2003). The 1986 USEPA epidemiological study found no correlation between illnesses and fecal coliform densities and has since 1986 recommended discontinuing the use of fecal coliform as a standard. In fact the 2001 USEPA TMDL guidance for pathogen TMDLs "strongly encourages" against using fecal coliforms to conduct TMDLs (USEPA 2001a, pg. 1-5). In conclusion, because it is based upon fecal coliform data, Dr. Teaf's indicator bacteria loading analysis has little relevance to actual illness risk to humans in the Illinois River Watershed.

5.5 THE PLAINTIFFS' CONSULTANTS MISREPRESENT PROBLEMS WITH INDICATOR BACTERIA

Due to the commonly acknowledged shortcomings associated with the use of the current indicator bacteria water quality criteria, the USEPA is actively working on updating these criteria, however the criteria remain unchanged since 1986 (USEPA 1986, 2004, 2007a). Indicator bacteria criteria are based on epidemiological data that are limited both in quantity and quality and are known to correlate poorly with pathogenic strains of bacteria and viruses (USEPA 2007a; NRC 2004). Furthermore, there is a substantial question whether indicator bacteria from non-human sources would have the same correlation to human illness rates as those from human sources (USEPA 2007a; NRC 2004). There are currently no studies that provide information about whether illness rates from fecal bacteria of animal origin would be similar to illness rates from fecal bacteria of human origin (NRC 2004, USEPA 2007a). In addition, many bacteria that may show up in fecal coliform tests are not fecal in origin, but are simply indigenous coliforms that can withstand high culture temperatures, and *E. coli* are not always fecal in origin, but in some cases can also be indigenous to the natural environment (NRC 2004; PBSJ 2000). These problems can result in indicator bacteria exceedances in the absence of substantial human sources (i.e., PBSJ 2000; McDonald et al. 2007). Unfortunately, revision of water quality criteria is not scheduled until 2012 (USEPA 2007b), so the current criteria will remain in use for some time, however they should always be considered in the context of these limitations.

The weakness in the indicator bacteria water quality criteria, while common knowledge, has been ignored or misrepresented by the Plaintiff consultants. For example, while Harwood et al. (2005) states

> *The failure of measurements of single indicator organisms to correlate with pathogens suggests that public health is not adequately protected*

Dr. Harwood in her May 2008 report states in paragraph 23.

> *The link between indicator bacteria concentration and human illness from recreational water use has been demonstrated in many epidemiological studies over the course of more than 50 years*

She further describes this historical epidemiological evidence in paragraphs 23 through 27 without once referring to the many commonly listed critiques of these studies and data. Dr. Teaf, in paragraph 17 of his May 2008 report, states

> *…microbial indicator organisms, as used in the context of impairment determinations, are commonly used and widely accepted measures of the potential for presence and health significance of pathogens, including bacteria, viruses and protozoa.*

Like Dr. Harwood, Dr. Teaf fails to acknowledge any of the common criticisms of the current bacterial water quality criteria, and his statement that they are widely accepted as "good indicators" is false. In contrast, the USEPA is very candid about the many shortcomings of these data and about the effectiveness of using these indicator bacteria groups for indicating illness (USEPA 1986, 2004, 2007a), and plans to commit substantial resources over the next few years to develop better methodologies for monitoring pathogen risk from water resource use (USEPA 2007b).

An example of further misrepresentation by Dr. Teaf can be found in paragraph 21 of his May 2008 report, with the following statement:

> *These indicator organisms, such as Escherichia coli, enterococci, and fecal coliform bacteria, may not cause illness directly, but they have demonstrated characteristics which make them reliable indicators of other harmful pathogens in water (Wade, 2006).*

Wade et al. (2006) contains no original indicator bacteria data using the standard indicator tests, and provides no opinion or information about the reliability of the indicators. Wade et al. (2003), is a well-known review of epidemiological studies relating illness rates to bacteria indicators. The combined analysis of 27 freshwater and marine studies in Wade et al. (2003) found no significant relationship between fecal coliform, and human illness, and a sometimes poor relationship between enterococci in fresh waters and human illness. Dr. Teaf does not cite Wade et al. (2003) in his report, but said he was aware of the results of Wade et al. (2003) in his deposition.

SECTION 6
THE WATER QUALITY IN THE ILLINOIS RIVER WATERSHED IS COMPARABLE TO OTHER WATERS IN OKLAHOMA

6.1 SUMMARY OF DETAILED FINDINGS

- The water quality of Lake Tenkiller is comparable to other systems within Oklahoma.
- Water quality of the rivers, specifically, Illinois River, is comparable to other rivers within Oklahoma.

6.2 THE WATER QUALITY OF LAKE TENKILLER IS COMPARABLE TO OTHER SYSTEMS WITHIN OKLAHOMA

Each year, the OWRB compiles a report detailing the state of water quality within Oklahoma's lakes and rivers (i.e., the BUMP report). In addition, every other year, Oklahoma is required by the USEPA to assess all waters of the state and determine which are not meeting their designated uses (e.g., fishable, swimmable, drinkable, etc.). Those not meeting their uses are called "impaired" and are required to undergo additional monitoring and analysis to determine what needs to be done to eliminate the impairment. These two water quality assessment exercises allow us to compare Lake Tenkiller's water quality to other reservoirs and lakes in the state.

The monitoring program for the BUMP tends to focus on water bodies that have potential water quality concerns and therefore, can result in a somewhat "biased" view of the water quality in the state. However, comparisons can still be made, while keeping this sampling protocol in mind. A review of the 2007 BUMP report provides a comparison of Lake Tenkiller's TSI with other sampled reservoirs and lakes (OWRB 2007). As discussed in Section 2, a TSI provides an estimate of the level of eutrophication in a lake, with higher numbers indicating greater eutrophication, in general. Figure 6-1 shows the chlorophyll-a TSIs for all lakes and reservoirs sampled from 2004 to 2007. These TSIs are representative of the summer (i.e., the BUMP sampling period) and include data from the entire water body (i.e., the BUMP assessment does not break out a reservoir into riverine, transitional, or lacustrine zones). Figure 6-1 indicates that 61% of the lakes sampled from 2004 to 2007 were classified as eutrophic or hypereutrophic, according to its chlorophyll-a TSI. Lake Tenkiller was one of those reservoirs, but 14% of the lakes were at a higher tropic level (hypereutrophic) than Lake Tenkiller. The probability distribution of the chlorophyll-a TSIs calculated from 2004 to 2007 shows that Lake Tenkiller lies at about the 58th percentile, meaning that about 42 percent of the lake's sampled had TSIs higher than Lake Tenkiller (Figure 6-2, bottom panel). In addition, the spatial pattern of chlorophyll-a TSI determined from Plaintiff's data collected in summer 2006 indicates that Lake Tenkiller's lacustrine area (represented by LK-01 and LK-02) is mesotrophic, which is typical for a run-of-the-river reservoir (see Horne 2009 for further discussion).

Inspection of total phosphorus collected during the BUMP effort shows a story similar to chlorophyll-a. Figure 6-3 displays the total phosphorus concentrations of the different reservoirs for summers of 2005 and 2007. Forty-percent of the lakes sampled during these two summers had phosphorus in the same range as Lake Tenkiller, while 37% had concentrations in a range higher than Lake Tenkiller.

Table 6-1 shows the biennial assessment of state waters from the preliminary 2008 report that was submitted to USEPA (ODEQ 2008). Only the constituents for which Lake Tenkiller is listed as "impaired" are shown in the table. Close to 11% of the assessed lakes are considered impaired based on chlorophyll-a and close to 63% of Oklahoma's assessed lakes are listed for low dissolved oxygen. Lake Tenkiller's chlorophyll-a impairment accounts for just 1.4% of the total assessed lakes and about 2% of the all the assessed lakes in relation to dissolved oxygen impairment. More importantly, Table 6-1 shows that there are many other lakes within Oklahoma that have water quality impairments. The assessment for total phosphorus is not yet performed on a state-wide basis and therefore, it is difficult to draw any conclusions regarding the impairment listing of Lake Tenkiller for total phosphorus.

Table 6-1. Percentage of lakes in Oklahoma with similar impairments as Lake Tenkiller.

Impairment	Size of Lake Tenkiller Impaired (acres)	Waterbodies in Illinois River Watershed		Lakes in Oklahoma		
		Total Lake Size Impaired (acres)	Total Lake Acres Assessed within Watershed	Total Acres of Lakes Impaired in Oklahoma (acres)	Total Lake Acres Assessed, with Sufficient Data or Information [2]	% of Assessed Lake Acres Impaired
Chlorophyll-*a*	8,440	8,440	14,034	66,222	622,176	10.6%
Dissolved Oxygen	13,470	13,470	14,034	389,498	622,176	62.6%
Total Phosphorus	8,440	8,440	n/a [3]	15,877	n/a [3]	---

Notes:

1. *Source: Oklahoma Department of Environmental Quality, 2008. The State of Oklahoma 2008 Water Quality Assessment Integrated Report.*
2. *Excludes 303(d) List Category 3 stream miles. Integrated report does not list acres assessed by impairment, only total acres assessed for any one constituent..*
   Category 3 - Insufficient or no data and information to determine if any designated use is attained.
3. *n/a = not available; lakes assessed for phosphorus unknown.*

The above information, combined with the analysis performed in Section 2.8 (i.e., the analysis of water quality in Lakes Hugo and Sardis watersheds) indicates that Lake Tenkiller's water quality is comparable to other reservoirs within the state. The water quality of Lake Tenkiller is not unusual and does not indicate significant issues. In fact, for a large portion of the lake (the lacustrine zone), the water quality is well within acceptable levels for chlorophyll-a and total phosphorus.

6.3 THE WATER QUALITY OF THE ILLINOIS RIVER IS COMPARABLE TO OTHER SYSTEMS WITHIN OKLAHOMA

6.3.1 Dissolved Oxygen Levels in the Illinois River Watershed Are Comparable or Better Than Many Other River Systems within Oklahoma

According to the State of Oklahoma's 303(d) list, low dissolved oxygen is a common problem in the state. About 2,500 miles of rivers and streams are listed as impaired for dissolved oxygen (Table 6-2). This represents about 20 percent of the total river miles assessed by the state. Within the Illinois River Watershed, the state listed only 1.6 miles as impaired due to dissolved oxygen and no part of the main stem of the Illinois River.

Table 6-2. Percentage of rivers/streams/creeks in Oklahoma with similar impairments as those in the Illinois River and its watershed.

Impairment [2]	Main Stem of Illinois River	All Waterbodies in Illinois River Watershed			Rivers/Streams/Creeks in Oklahoma		
	Total Stream Miles Impaired	Total Stream Miles Impaired	Total River Miles Assessed [3]	% of Assessed Stream Miles Impaired	Total Stream Miles Impaired	Total River Miles Assessed, with Sufficient Data or Information [4]	% of Assessed Stream Miles Impaired
Dissolved Oxygen	0	1.6	551.5	0.3%	2,547	12,511	20.4%
Enterococcus	12.9	112.2	551.5	20.3%	6,977	12,511	55.8%
Escherichia Coli	31.7	37.9	551.5	6.9%	3,495	12,511	27.9%
Fecal Coliform	31.7	31.7	551.5	5.7%	3,094	12,511	24.7%
Lead	31.7	31.7	551.5	5.7%	1,437	12,511	11.5%
Total Phosphorus	60.2	92.8	92.8	100.0%	160	185 [5]	86.5%
Turbidity	5.2	5.2	551.5	0.9%	4,012	12,511	32.1%

Notes:

1. *Source: Oklahoma Department of Environmental Quality, 2008. The State of Oklahoma 2008 Water Quality Assessment Integrated Report.*
2. *Only impairments listed for the main stem of the Illinois River are listed.*
3. *Appendix B of integrated report does not list miles by impairment. Assumed that miles reported pertain to all constituents except phosphorus.*
4. *Excludes Category 3 stream miles. Integrated report does not list miles assessed by impairment, only total miles assessed.*
   Category 3 - Insufficient or no data and information to determine if any designated use is attained.
5. *Total river miles estimated from 'Scenic River'-designated water bodies in Oklahoma.*
   Estimated based on Scenic River area descriptions in Oklahoma Statute.
   Length of Big Lee's Creek not limited by the 420-foot MSL elevation due to limited available information.

Using data collected between 2004 and 2007, I looked at dissolved oxygen conditions throughout the state.[26] Many locations failed to meet the dissolved oxygen standards[27] and a number of locations had problems in multiple years (Figure 6-4). In contrast, within the Illinois River Watershed only one small section of river did not meet standards, and that was only during one of the four years considered.

[26] Only locations with at least eight records in at least two years were considered. In addition, to ensure year-round oxygen status, only locations with at least one dissolved oxygen records in at least three quarters (three-month periods) were considered.

[27] The Oklahoma dissolved oxygen regulations are written such that if 10% of readings at a particular location are found to be below a certain criteria, that location is considered impaired due to low dissolved oxygen. In the summer in the much of the Illinois River Watershed, that the criteria are 5.0 mg/L., and 6.0 mg/L for the rest of the year. In some other areas of the Oklahoma the summer and rest-of-the-year the criteria are 4.0 mg/L and 5.0 mg/L, respectively.